

The use of Biofiltration cells to Filter Contaminated Water flowing from a Slum Settlement in South Africa

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Geographical Science

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Abstract

Polluted urban surface runoff degrades the receiving water bodies and impacts on downstream water quality and ecological systems. In response, there is growing research attention that is focused on how to treat surface water runoff before it is discharged into these water bodies which includes using a variety of land-based treatment systems. This thesis investigates the performance of large scale, low-cost nature-based filtration systems to clean contaminated water without the addition of chemicals. A relatively small portion of water that is generated and discharged from a slum settlement in South Africa, where water-based services are limited and often dysfunctional, is intercepted and diverted through six biofiltration cells. These cells were packed with different types of natural media, three of which were planted with a variety of reeds while the other cells were kept as control cells. Water that flows into each biofiltration cell is controlled via a network of valves. Flow meters were used to determine the volume and rate of discharge to each cell. The purpose of this study was to determine the effects of HLR (hydraulic loading rate) and HRT (hydraulic retention time) on water quality that was discharged from each cell. This study determined whether the resulting effluent could be repurposed for irrigating edible crops. The final discharge was tested to confirm the differences between the influent and effluent in each cell. Overall the vegetated cell that was packed with large stones (19 – 25 mm aggregates) (LSV) performed the best and displayed reductions of 98.51% of ammonia and 100% of orthophosphate concentrations. *E. coli* bacteria were also reduced by nearly 100%. Phytoremediation played a role in reducing contamination by removing 97.07%, 89.70% and 100% for ammonia, orthophosphate and *E. coli* respectively over the study period of four months. Throughout the study, Large Stone Vegetated cells (LSV) reduced nitrite levels by 77.21% with higher removal rates for ammonia, orthophosphate, nitrites, respectively, compared to Large Stone cells (LS). An HRT of approximately seven days resulted in the most improved water quality for LSV, LS, Small Stone (SS) and Small Stone Vegetated cells (SSV) for most of the parameters that were tested. However, orthophosphate leaching occurred in the SSV cell. Peach Pip Vegetated cells (PPV) and Peach Pip cells (PP) did not perform as well as the other cells.

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Chapter 1: Introduction

1.1 Rationale

Contaminated surface water runoff from slum settlements negatively impacts on the quality of the receiving environment and adversely affects the health and well-being of the citizens living in the settlement (Jamwal *et al.*, 2011; Capps *et al.*, 2016). This study aims to assess and treat surface runoff that is discharged from a slum settlement and reuse it safely for irrigating edible crops. Drainage facilities and sanitary conditions in slum settlements often comprise of ad hoc arrangements that are poorly managed and dysfunctional (Ajibade *et al.*, 2013). In South African slum settlements this contaminated surface runoff is usually a mixture of blackwater, greywater, stormwater and solid waste which is usually discharged directly into surface water bodies (Armitage, 2011). The contaminated surface runoff includes diverse pollutants such as nutrients, pharmaceuticals and faecal bacteria (Joshi *et al.*, 2014; Katukiza *et al.*, 2015). In addition, impervious surfaces reduce the potential for infiltration resulting in an increase in runoff and concentrated surface pollutants (McKee *et al.*, 2003).

The effects of poor drainage from slum settlements on ecosystem services and impacts on biodiversity are largely unexamined (Capps *et al.*, 2016). While hydrology and water quality studies are more extensively researched in urban areas in developed countries (Butler & Davies, 2011; Rodríguez *et al.*, 2013), surface water runoff and water quality flowing from slum settlements is poorly understood. The goal of this study is to improve knowledge and understanding of pollutants in surface runoff and how this water could be cleaned by using nature-based processes to reduce contamination for safe re-use of the water. The rationale for this study is to establish whether the resulting effluent from the biofiltration cells could be reused for irrigating urban agricultural gardens in compliance with the guidelines of the Department of Water and Sanitation in South Africa.

1.2 Aims and Objectives

The study analyses the performance of biofiltration cells to filter and treat polluted water from a slum settlement. The performance will be measured while varying the hydraulic retention time (HRT) and hydraulic loading rate (HLR) in a scientific experiment conducted at a demonstration site in close proximity to a slum settlement. HLR and HRT are used to manipulate the flow and quantity of water during the experiment. The goal is to determine whether the effluent can be repurposed urban agricultural use. The aim will be achieved by the following objectives:

- To determine the water quality of the influent. In this objective the polluted water from the settlement that flows into the stream that passes through the study site will be tested and the results compared to the effluent that is released from each of the biofiltration cells.
- To examine whether the treated water is safe to discharge into a freshwater system and also for re-use for irrigating edible crops.
- To determine how HLR and HRT affects the performance of the biofiltration cells to treat water within the respective cells.

HLR and HRT are key variables that are known to affect the treatment efficiencies of wetland systems (Lu *et al.*, 2009). Tanner (1995) reported that longer HRT resulted in greater reductions in phosphorous levels. In addition, lowering HLR ensures a longer HRT, thereby improving denitrification and reducing nitrogen levels (Liu *et al.*, 2014). Yang (2001) also examined longer HRT and lower HLR resulting in a reduction of contamination levels. However, nature-based material in the substrate of wetlands or filter systems and the quality of the influent contamination greatly affect HRT and HLR efficiencies (Yang *et al.*, 2001).

1.3 Background

Problems arising from contaminated runoff are creating environmental damage worldwide including South Africa (Dai, 2011). Interest from the research community is growing in discovering techniques that integrate existing infrastructure with surface runoff controls to diminish water pollution and increase water conservation (Chocat *et al.*, 2001). The goal is to build sustainable urban practices that optimize water treatment in general and improve environmental conditions for communities, if possible, without using chemical additives.

Chemical and biological water treatment is practised worldwide. Biological treatment at waste water treatment plants, for example, is rarely achieved without the addition of chemicals such as chlorine. Alternatives exist, such as reverse osmosis, but these are expensive processes involving microfiltration, solvent extraction and ion exchange (Gupta *et al.*, 2012). Trihalomethanes are formed while disinfecting water with chlorine and other chemical disinfecting agents. The trihalomethanes that are formed from chlorination are known to be carcinogenic and may be responsible for increasing the rate of cancer in human populations (Hsu *et al.*, 2001). The challenge that is creating interest broadly for environmental engineering sector is develop nature-based solutions that are safe and economical in treating contaminated runoff, in this instance from urban areas that have poorly developed infrastructure (Srivastava & Majumder, 2008). Biofiltration techniques are known to increase pollutant removal using natural media as filter material and in

the management of HLR and HRT (Yang *et al.*, 2001). Wetlands have been utilized to treat contaminated wastewater and reuse the treated water to grow agricultural crops (Cirelli *et al.*, 2007).

1.4 Overview of Literature

One of the most critical elements in determining the efficacy of biofiltration systems is the filter media and biomass. Biofiltration methods are considered economical and produce improved water quality. Cells have a long operational life and are considered simple to operate and maintain (Chaudhary *et al.*, 2003). According to Yang (2001) biofiltration systems have proven application in the aquaculture sector whereby water can be treated to required standards with an HRT of 2.5 hours. Bratieres (2008) also concluded that biofiltration systems can be used to remove nutrients, heavy metals and pathogens from polluted water. Variables that can affect the treatment performance include depth and type of filter media; vegetation as a phytoremediation measure; HRT and HLR.

Clogging is one of the most important factors that determine the long-term suitability of a biofiltration system. However, it is less likely to occur when the appropriate HLR is applied (Loudon & Birnie, 1991). Using longer HRT has shown higher removal of contaminants was observed in a constructed wetland with a longer HRT and has also been shown to make a biofiltration system more robust against toxic substances (Hoffmann *et al.*, 2011).

1.5 Overview of Research Design and Methods

The study site is located the Water Hub on the outskirts of the town of Franschhoek, Western Cape, South Africa. It is situated 1 km downstream of slum (informal) settlement of Langrug. As mentioned previous, the study aimed at understanding the capabilities of biofiltration systems in treating highly contaminated surface runoff. This study entailed testing various substrates and vegetation in a controlled environment.

Six wastewater treatment drying beds were restored and retrofitted at the Water Hub into biofiltration cells. Three cells were planted with indigenous vegetation and three cells were constructed without vegetation for experimental purposes. Each of the vegetated cells were planted with *Phragmites australis*, *Typha capensis* and *Cyperus textilis*. These species were chosen for their ability to clean contaminated conditions (Milandri *et al.*, 2011).

The substrate material in each paired cells, that is vegetated and non-vegetated cells, were filled with large stones, small stones and peach pips respectively. Peach stones were used as medium for

the biofiltration cells and were referred to as peach pips in this study. Adequate loading rates and retention time was applied to achieve optimal treatment efficacy. HLR and HRT have a linear relationship. The study aimed to assess how HLR and HRT affected water quality in each biofiltration cell.

1.6 Study Site: The Water Hub

The formal town of Franschhoek is a small urban area located approximately 100 km north of Cape Town. The slum (informal settlement is the term used in South Africa) settlement known as Langrug is situated approximately 1 km upstream of the Water Hub. Polluted water enters the Water Hub via the Stiebeuel River (Figure 1). The contaminated water from Langrug flows into the Stiebeuel River which bisects the study site before flowing into Franschhoek River on the western boundary of the site.

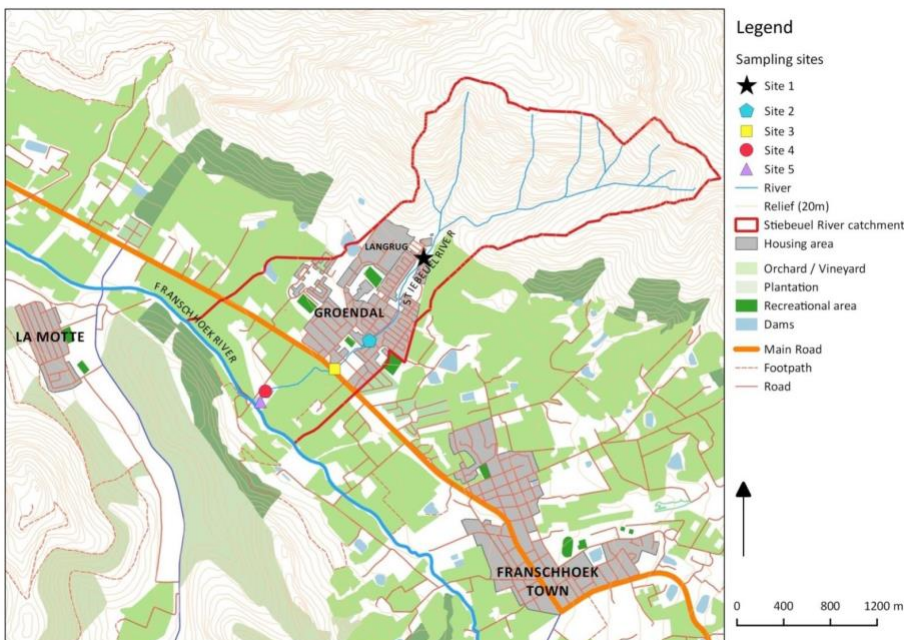


Figure 1: Map of the Langrug study site in relation to Franschhoek (Fell, 2018)

The Water Hub research and demonstration site received seed funding from the Western Cape government and is in a partnership with the Stellenbosch municipality, the land owners. The site, which once operated by the municipality as the Franschhoek Waste Water Treatment Works (WWTW), was abandoned in 2013 after the sewerage system was diverted to a new plant approximately 8km away. The old Franschhoek WWTW had reached its capacity in dealing with an increasing volume of effluent from 2006 or even earlier. Temporary arrangement were made

on the site from 2010 to 2013 while the new plant was being constructed. In 2013 old WWTW was abandoned by the municipality until 2016 when the provincial government decided that the site could be used as a centre for research in Sustainable Urban Drainage (SuDs). Accordingly the provincial government procured the services of a small consulting engineering company, and in partnership with the University of Cape Town, with the request to draft a conceptual plan for site and also provided a small amount of capital to develop and retrofit the infrastructure in accordance with the plan (www.thewaterhub.org.za).

The Stiebeuel River is located alongside the slum settlement. The population of approximately 6,000 residents live in makeshift houses built from an assortment of corrugated iron, wood and other available materials including used doors and windows. The slum only has 40 flush toilets in all, many of which are dysfunctional. Surface water from the settlement is generated from a combination of discarded water including that from communal washing facilities, public taps and dysfunctional leaking toilets. Contaminated water is discharged into the Stiebeuel River (Fell, 2018) which is used as water source in this study and is referred to as the influent. In the experiment, water was abstracted from the Stiebeuel River to two 10 000L storage tanks and conveyed later into the various drying beds.

1.7 Limitations

(a) Time constraint

Natural remediation methods take time to become established. For example, it requires time to fully understand the contribution of phytoremediation to treat contaminated water (Cunningham *et al.*, 1997). In addition, microbial activity requires more time to populate the biofiltration cells (Tao *et al.*, 2006). The full potential of these systems may only be observed over a longer duration.

(b) Microbial Growth

Previous studies have reported that microbial activity is linked to temperature (Atlas & Bartha, 1981; Faulwetter *et al.*, 2009). Bacterial growth and metabolic rates diminish as temperature decreases. Thus, winter is not the most suited time for microbial growth. This could affect the data from this study as all the samples were taken during the winter. According to Werker (2002), denitrification was only detected when temperatures were above 5 °C and nitrification activity is observed between 6-10 °C. The Franschhoek valley resides in a mild Mediterranean climate with

cooler conditions during the winter, but there were periods when ambient air temperatures were below 5 °C.

Chapter 2. Literature Review

Over 30% of urban populations in developing countries reside in slum settlements (Mahabir *et al.*, 2016) . These settlements lack planning guidelines, regulations and access to adequate drainage facilities (Parkinson *et al.*, 2007). Rapid urban growth has led to slum settlements being built around urban areas on land that would previously have been designated for development purposes or deemed inadequate for construction due to environmental or physical factors (Parkinson *et al.*, 2007). Slums are characterised by high population densities and widespread poverty. They are also at risk of flooding due to the occupation of low lying land often deemed unsuitable for housing development and the lack of drainage.

Contaminated runoff from slums includes a mix of blackwater, greywater and solid waste (Armitage, 2011) that has potential to pollute surface water bodies, degrades aquatic systems and natural habitat, and places downstream users at risk. It is mainly for these reasons that the study aims to investigate an option to treat runoff and improve knowledge and understanding about how water can be cleaned or polished for reuse using nature-based processes.

2.1 Biofiltration

Biofiltration designs include constructed wetlands, vegetated filter strips and bioswales (Jurries, 2003). This study concentrates on understanding the form of biofiltration cells as constructed wetlands. Wetlands provide services that include groundwater recharge, flood management, nutrient cycling and flow control, sometimes referred to as “the kidneys of the landscape” (Acreman *et al.*, 2003). The use of wetlands for wastewater treatment were first carried out by Dr. Kathe Seidel in the 1950’s at the Max Planck Institute located in Plön, Germany. Seidel built full scale wetland systems in the 1960’s, where she grew macrophytes in shallow embankment ditches (Vymazal *et al.*, 2006). Constructed wetlands have also been used to treat agricultural/industrial wastewater, storm water runoff and leachate from landfills (Vymazal, 2014). Wetland systems have shown the ability to remove contaminants or capture particles by filtration, sedimentation, adsorption, uptake by vegetation, chemical precipitation and microbial activity (Kivaisi, 2001).

2.1.1 Biofiltration Cells in the form of Constructed Wetlands (CW)

Any type of filter that has biomass attached to it can be considered a biofilter. Thus, filtration via rocks or sand filters can be considered a form of biofiltration. The fundamental concept of biofiltration involves microorganisms that are attached to the filter media. These microorganisms biodegrade contaminants present in the medium (Chaudhary *et al.*, 2003). Biofilters are capable of reducing total organic carbon (TOC) (Kumar *et al.*, 2013). In addition to removing and reducing heavy metals and TOC, biofilters assist in nutrient removal and are efficiency in removing total

suspended solids (TSS) (Bratieres *et al.*, 2008). Biofilters are also efficient in lowering BOD and coliform bacteria (Jowett & McMaster, 1995).

Research interest is also growing to determine the extent to which biofilters are capable of treating residuals from personal care products. A study has shown various forms of biofilters are capable of removing a wide range of pharmaceutical products by up to 90% from a waste water stream (Reungoat *et al.*, 2011). According to a study by Lee (2012), biofiltration methods and designs could potentially treat pharmaceuticals and personal care products in wastewater more efficiently than reverse osmosis. However, Chen (1994) claims that biofiltration methods need to be directly linked to HRT when the residency time is at least 15 days.

Biofiltration cells should be designed to mimic natural treatment processes and involve soils, flow, vegetation and the microbial activity in the treatment. Combining these natural technologies takes advantage of biological, chemical and physical processes that occur in controlled wetland environments (Wu *et al.*, 2013). Previous studies have shown that COD, BOD, SS, nitrogen, phosphorous and bacteria can be treated via constructed wetland (Vymazal, 2007). Microbial nitrification and denitrification are responsible for nitrogen removal in constructed wetlands (Brix & Schierup, 1990; Hammer & Knight, 1994; Tanner *et al.*, 1999). The efficiency of nutrient removal varies according to the type of vegetation, quality of wastewater, treatment capacity of filtration cells, substrate, HLR and HRT. Substrates should have a high adsorption capacity, in which the vegetation should have extensive nutrient accumulation and assimilation abilities and in which the HRT should be given sufficient time to adsorb or remove contamination (Wu *et al.*, 2013).

2.1.2 The Benefits of Constructing Biofiltration Cells

Typically biofiltration cells vegetated constructed wetlands (CW) and engineered systems that have been designed to use natural procedures in the treatment of polluted water. The processes involve microbial activity, vegetation and substrate form. When combined these processes provide water treatment capabilities. CWs are designed to mimic the same processes that occur in natural wetlands, but in a controlled environment (Vymazal, 2010). CWs are also used in the reuse of water in agriculture; in improving environmental aesthetics; and in habitat restoration (Kivaisi, 2001). The advantages of constructed wetlands is that they are cheaper compared to conventional treatment infrastructure, easier to operate and more cost effective to maintain (Parkinson & Taylor, 2003). Despite these advantages, Kivaisi (2001) found that CW systems were extensively used in the developing world

According to a study by Molle (2005) constructed wetlands have reduced suspended solids (SS) by 95% and nitrification rates by 85%. Constructed wetlands incorporate both vertical and horizontal flow systems that are capable of removing more than 90% of organic load. Both types of flow systems were observed to reduce nitrogen and phosphorous (Luederitz *et al.*, 2001). However, the removal of both P and N is usually most efficient when combined with lower HLR. As loading rates increase, removal efficiency decreases. Large areas of land are usually required to increase efficiencies in removing P and N (Nichols, 1983).

2.1.3 Biofiltration limitations

One of the largest drawbacks in the use of biofiltration cells is the land requirement. Large volumes of water will require large biofiltration cells to be constructed (Kumar, 2013). According to Kumar (2013) biofiltration units can require long assimilation periods for microbial populations. Microbial growth is imperative for the biofiltration cells to effectively treat polluted water. Thus, a longer duration may be required to discern the biofiltration cells full treatment potential.

2.2 Horizontal Subsurface Flow Constructed Wetland

In horizontal subsurface flow (HF) wetlands, the contaminated water flows horizontally through the substrate that is planted with suitable vegetation as displayed by Figure 2. This version of CWs can be relatively inexpensive and can also be maintained easily as they do not require technical expertise. Thus, they provide various advantages such as low operation costs and can be built using local materials (Hoffmann *et al.*, 2011). This flow regime has been shown to reduce BOD (biological oxygen demand) and pathogens requiring minimal maintenance. HF systems also do not require energy and thus are a cleaner form of filtration. However, they require a large area of land and there is a risk of the systems being clogged (Knowles *et al.*, 2011). This form of constructed wetlands also requires expert design and construction to ensure that the systems can function appropriately.

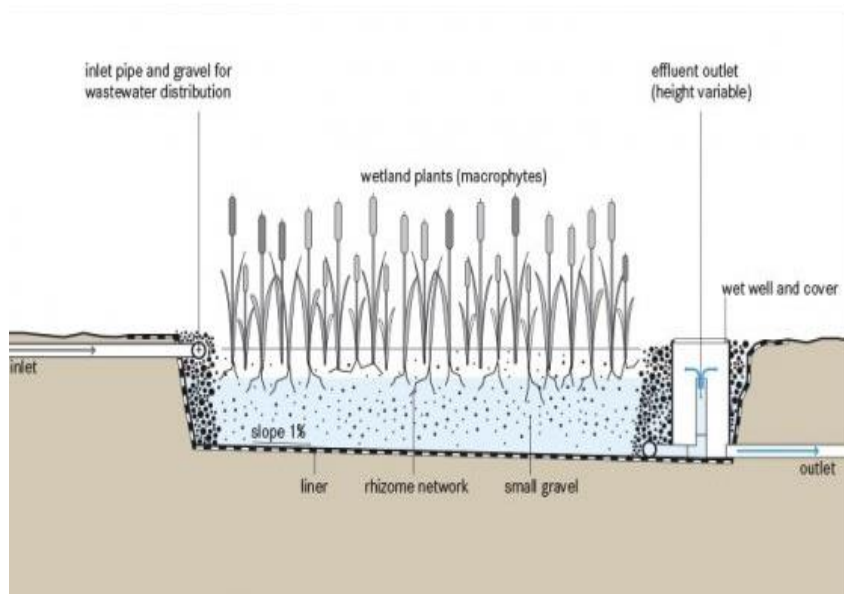


Figure 2: HF wetland (Tilley, 2008)

2.3 Vertical Flow Constructed Wetland

Vertical flow (VF) systems involve pouring contaminated water from above via a mechanical system as displayed by Figure 3. Polluted water is applied via a pump or a syphon from where it flows vertically down through the substrate to the bottom of the bed and is collected by drainage pipes. Clogging can become an issue in VF systems and a pre-treatment of the influent is required. This process separates solids from the contaminated water (Hoffmann *et al.*, 2011). VF systems have certain advantages over horizontal flow systems. They require less space and the risk of clogging is also lower than HF systems. VF systems also provide good aeration conditions (Mcbride & Tanner, 1999). However, VF systems require energy and more frequent maintenance than HF systems. VF systems also require expert design and construction. VF systems also require high quality filtration material (Morel, 2006).

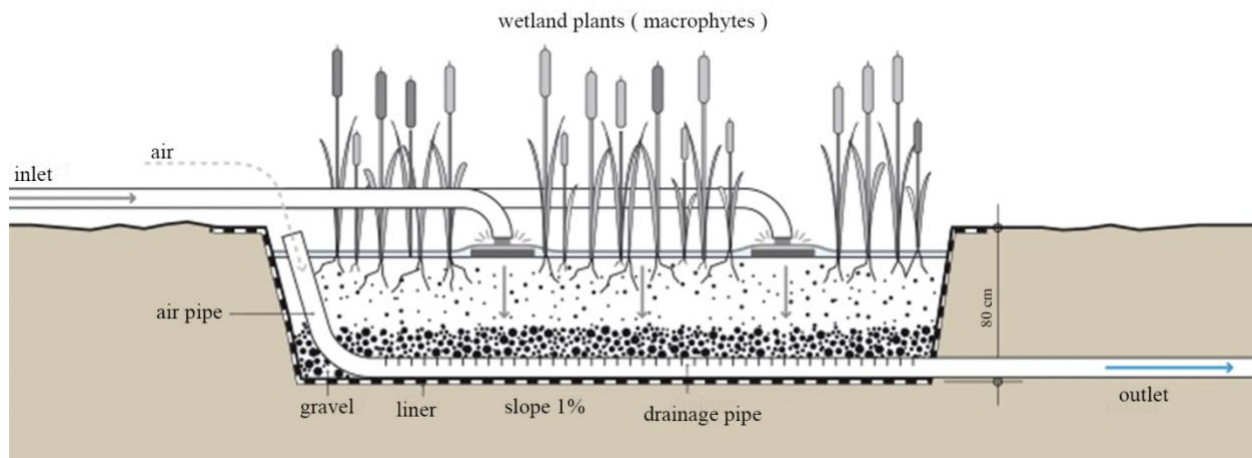


Figure 3: VF wetland (Tilley, 2008)

2.4 Hybrid Constructed Wetland

Different flow regimes can be used to achieve a higher treatment efficiency. Most hybrid systems involve combining horizontal and vertical filtration together as displayed by Figure 4. HF and VF systems provide different benefits and when combined have the potential to provide benefits of both systems. Hybrid systems have a higher filtration efficiency when it comes to nitrogen. They require minimal electricity (pumps or syphons) and can provide employment to local citizens of the area (Vymazal, 2005). However, hybrid systems require a large area and expert design and construction. If land is available, then hybrid systems have the potential to filter polluted water for larger communities in suburban areas (Lipkow *et al.*, 2010). They also require pre-treatment of solids to reduce the risk of clogging. High quality filter material is required for these systems and are not always available. Thus, hybrid CWs can be expensive (Morel, 2006).

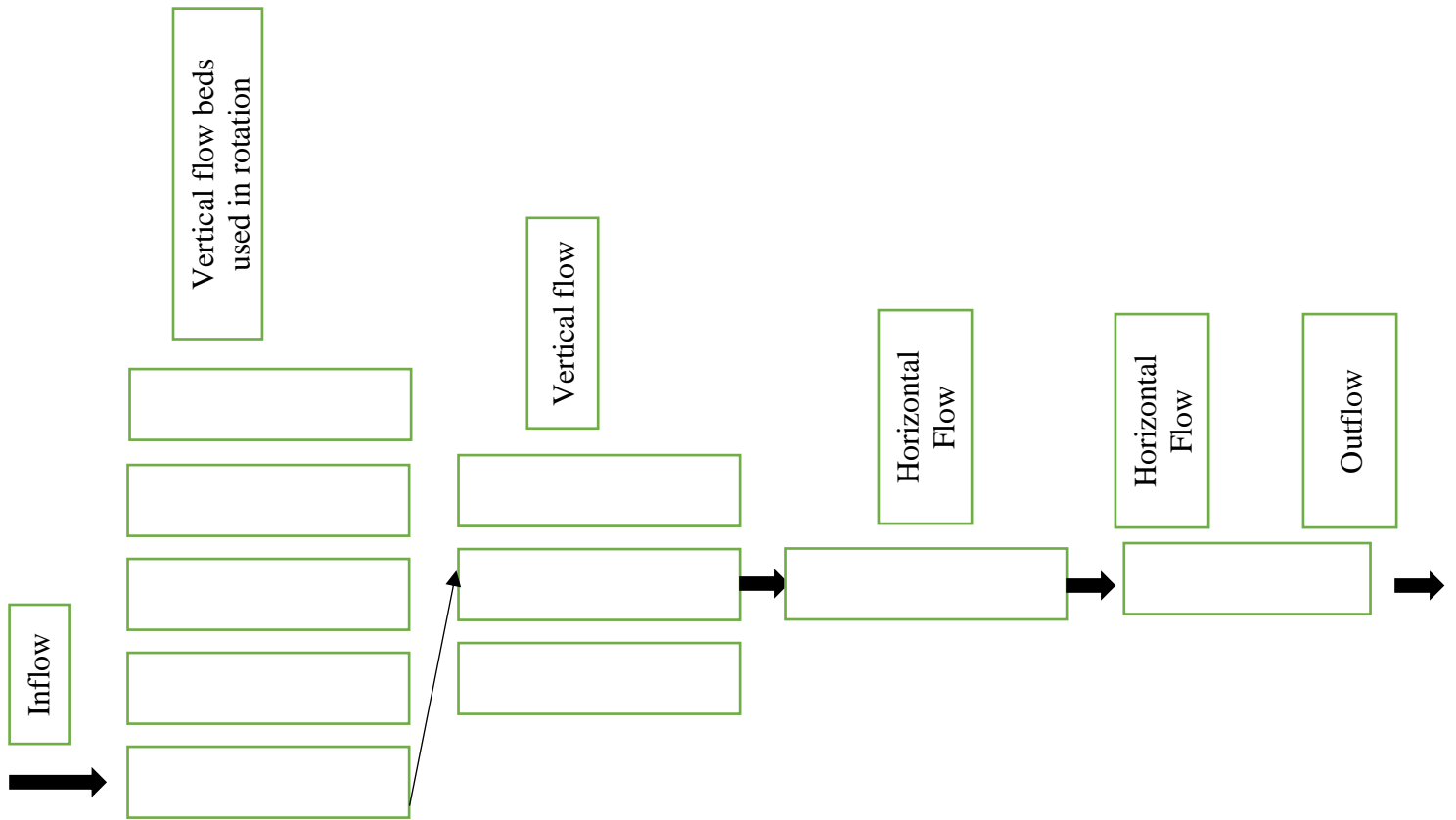


Figure 4: Hybrid Wetland (Vymazal, 2005)

2.5 Free Water Subsurface Constructed Wetland

Free water subsurface (FWS) constructed wetlands entails a succession of filtration sequences via planted channels as displayed by Figure 5. In these systems, polluted water flows above ground level and the plants are established via their root systems at the base of the bed. The plants can also float in the water. The aim of FWS systems is to mimic natural wetlands or swamps. The drying beds are lined with an impervious barrier and covered with gravel, rocks and soil. Native vegetation is usually applied to these systems. FWS are visually pleasing and provide a habitat for animals and promote biodiversity (Tilley, 2008). They have low operating costs and can be constructed with local materials, thus not requiring high cost filtration materials (Hoffmann *et al.*, 2011). No energy is required for these systems and they can be combined with agriculture and aquaculture. However, they require large spaces, expert design and construction and supervision. Without these considerations, an FWS system could fail (Tilley, 2008).

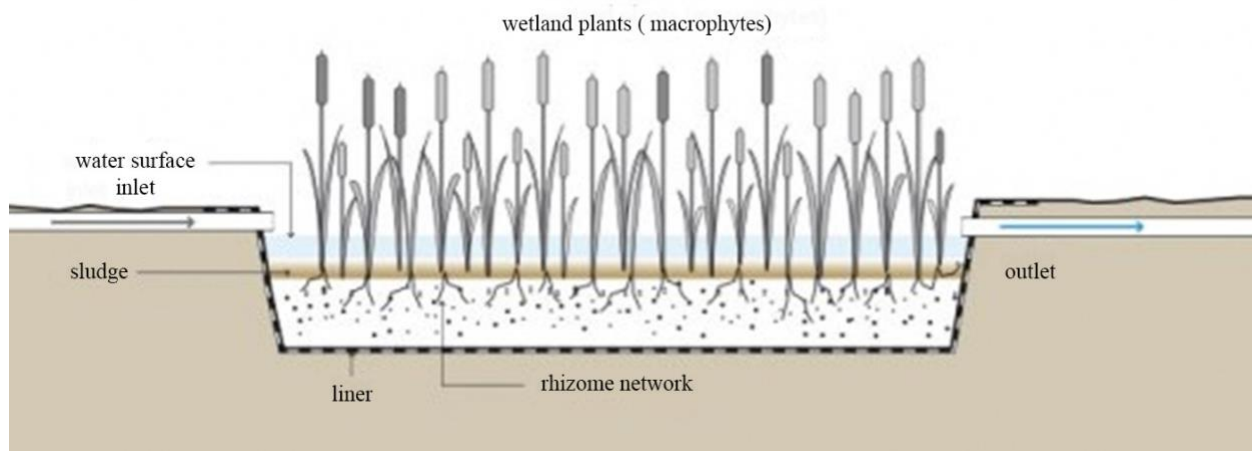


Figure 5: FWS wetland (Tilley, 2008)

2.6 Subsurface Biofilters

A subsurface biofilter involves flowing polluted water through a media system that usually consists of sand or gravel. They are also known as non-planted filters or percolation beds. These systems have been implemented in cold climates for domestic greywater filtration. Subsurface biofilters are designed similarly to constructed wetlands and have the option of utilizing VF, HF and FWS systems. Subsurface biofilters require the polluted water to be pre-treated before it enters the system. The filtered water infiltrates through the substrate until it percolates into the underlying soil (Morel, 2006). According to Morel (2006), subsurface biofilters can reduce BOD, TSS and TN (Total Nitrogen) levels. However, these systems pose a high risk of clogging and require high quality substrate material. Subsurface biofilter may require electrical pumps and need to be expertly designed and constructed. Without these considerations, these systems could potentially fail or not adequately filter the polluted influent.

2.7 Phytoremediation

Phytoremediation is defined as using living plant technologies to purify water, air and soil that is contaminated with hazardous waste (Reichenauer & Germida, 2008). It is an emerging technology that uses plants and the associated microbial activity that occurs in the rhizosphere to filter toxic substances in sediments, soil and water bodies including groundwater. Phytoremediation has been used for treating pesticides, heavy metals and landfill leachate (Susarla *et al.*, 2002). Phytoremediation is an economic method towards rehabilitation of the environment due to the

plants ability to metabolize concentrated compounds from their environment into their tissues. This technology involves utilizing certain plants known as hyperaccumulators that can purify pollutants in soil and water. Organic pollutants and heavy metal contamination are key targets for filtration via phytoremediation (Salt *et al.*, 1998). Hyperaccumulator plants can adapt to soils with high metal toxicity and absorb these pollutants in their roots. The roots from these plants can extract metals from the soil quicker and convey the contaminants swiftly to their shoots and store large quantities in their leaves as well. This process returns degraded ecosystems to a less hazardous state (Rascio & Navari-Izzo, 2011).

Plants also provide surface area for microbial activity and thus increase their systems treatment capabilities. The surface area provided can lead to microbial growth. The biofilm that is created by the bacteria is accountable for most of the microbial activity that occurs in constructed wetlands (Decamp & Warren, 2000). Contact between the polluted water and the macrophytes in the wetland are essential for the alleviation of agricultural pollutants that contain a variety of nutrients (Wu *et al.*, 2013). Macrophytes also play a large role in transporting oxygen located in the rhizosphere. Studies have shown that the oxygen transported is approximately 90% (Brix, 1997). The transport of oxygen stimulates the growth of nitrifying bacteria and the aerobic decomposition of organic matter (Lee & Scholz, 2007). A major factor that determines the success of constructed wetlands is clogging. One of the main functions of plants is to counteract clogging of the substrate (Brix & Arias, 2005). According to Decamp (2000) Macrophytes provide treatment and can have a beneficial effect on *E. coli* removal.

Plants are able to ameliorate organic contaminants via direct uptake of pollutants and accumulate them into plant tissues while releasing enzymes and exudates that encourage microbial activity. Plants also offer an additional benefit of increasing organic carbon in the soil which further encourages microbial activity. Plants with deep roots also reduce soil erosion by stabilizing and binding the soil (Schnoor *et al.*, 1995). The presence of macrophytes has been shown to provide additional treatment abilities to remove *E. coli* bacteria (Decamp & Warren, 2000). Studies have shown that using polycultures (using more than one species) may perform better than traditional monocultures (using one species). Polycultures have shown to treat wastewater more efficiently, but according to Coleman (2001) this also depends on the species and its treatment capabilities.

2.8 Hydraulic Retention Time (HRT)

Hydraulic retention time can be defined as the ratio between flow rate and volume of the surface water in the biofiltration cells. Thus, HRT can be increased by decreasing volume or increasing water depth (Toet *et al.*, 2005). According to Toet (2005) increasing HRT resulted in meeting

desired bathing water standards for ammonium and faecal coliform. However, seasonal changes affect HRT. Toet's study goes on to mention that bathing standards were met with an HRT of 4 days for most of the year. Conversely, an HRT of 0.8 days was sufficient to remove faecal coliform and ammonium during the spring and summer period, using an HLR of 50 litres per day. The study further mentions that substantial phosphorous removal will only happen with an HRT of 15 days or more. Tao (2006) mentioned that it can take <1-6 weeks for the maturation of microbial activity for the biofilm on the submerged plant surfaces. High reduction efficiencies were reached by HRTs of up to 25 days or less. The treatment performance can also vary according to temperature, season and influent strength. Previous studies have demonstrated that contaminated water can be filtered for nitrogen, phosphorous, organic matter and SS with an HRT ranging from 1-12.8 days (Schwartz & Boyd, 1995; Lin *et al.*, 2003; Lin *et al.*, 2005). Longer HRT plays a significant role in removing COD, BOD, over 90% of nitrates and TKN in conjunction with removing 100% of ammonium (Ghosh & Gopal, 2010). However, treatment performance can vary with HRT and HLR. Garcia (2004) tested drying beds that performed better with HRT of 2.5-5.5 days rather than longer HRT of 4.5-10 days. Studies have shown higher performance nutrient removal with higher HLR and lower HRT (El-Bestawy *et al.*, 2005). According to El-Bestawy (2005) greater removal efficiencies were seen with higher HLR and shorter HRT. Thus, this study will assess the required HRT needed to achieve adequate water treatment. This data can then be utilized to determine the activities the effluent should be repurposed for such as urban agriculture. Insuring water security is of the utmost importance in every society and repurposing water for urban agriculture is one of the best ways to do so.

One of the most important factors in this study is to determine the hydraulic retention time required to treat the polluted water. HRT has exhibited that it can influence various water quality measures. There is a direct relationship between hydraulic retention time and *E. coli* removal in biofiltration cells (Netter, 1993; Decamp & Warren, 2000). High hydraulic retention time of up to 13 days has exhibited the removal of organic contaminants in constructed wetlands (Masi *et al.*, 2002). According to Lee (2007) HRT can be reduced if the inflow and outflow rates remain constant. This would increase the efficacy of the biofiltration cell and make the system perform more efficiently. High retention time has also shown to be more robust against toxic substances compared to systems that do not allow sufficient retention time (Hoffmann *et al.*, 2011). Previous studies have stated that HRT is one of the most important parameters in judging the performance of a wetland system (Kadlec & Knight, 1996; Dong *et al.*, 2011).

2.9 Hydraulic Loading Rate

Hydraulic loading rate is defined as the rate at which polluted water enters the soil (Eliasson, 15

2002). Previous studies have shown that increasing the volume of influent and reduced retention time reduces the treatment efficiency of a system (Dong *et al.*, 2011). HLR is one of the main considerations taken into designing systems that filter contaminated water. HLR is estimated by either soil analysis or percolation tests (EPA, 1980). There are various factors used to determine the HLR of a system. Soil structure, soil aeration, bulk density and effluent quality can impact HLR. Soil morphology is considered the best method to determine the infiltration capacity of the soil and determine durability of the media. It is paramount to ensure that the infiltration rate of the water is lower than the HLR. If, this does not occur than the system can succumb to hydraulic failure and soil clogging (Eliasson, 2002). The rate and severity of the clogging depends on the HLR, influent quality, temperature, soil moisture and aeration status (Loudon & Birnie, 1991). According to Eliasson (2002), soil clogging can be reduced by decreasing HLR and ensuring that the influent levels of TSS and BOD are not above a given level. There is a direct correlation between BOD and HLR (Tyler *et al.*, 1995). According to Tyler (1995) polluted water with low BOD levels can be applied at 2-16 times the usual HLR. Thus, pre-treatment of an influent for BOD could result in being able to filter a much larger volume of water. Therefore, the quality of the influent and the soil morphology plays a role in determining the optimum HLR.

HLR can have a variety of effects on a biofiltration system. According to Endut (2010) increasing the HLR increased plant production and can be implemented to attain suitable conditions for fish growth as well. Previous studies have stated that HLR is one of the most important parameters in judging the performance of a wetland system (Kadlec & Knight, 1996; Dong *et al.*, 2011). HLR is calculated by the formula of $(Q \times 100)/(A)$, where Q equals the flow within the wetland system and entails precipitation, infiltration and evapotranspiration. (A) equals the total surface area of a pond (Dong *et al.*, 2011). This is one of the flaws of wetland systems as they may require large land area to treat greater volumes of polluted water (Lin *et al.*, 2005). Previous studies have demonstrated that contaminated water can filtered for nitrogen, phosphorous, organic matter and SS with an HLR ranging from 18 mm-135 mm/day (Schwartz & Boyd, 1995; Lin *et al.*, 2002; Lin *et al.*, 2005). The maximum HLR that can be attained without surface flooding are affected by factors such as biofilm growth, TSS loading rate, media size and distribution (Cooper, 2005). Previous studies have indicated that lowering HLR improves nutrient removal efficiency and the filtration of organic matter (Mæhlum & Stålnacke, 1999; Garcia, 2004). However, El- Bestawy (2005) also mentioned that greater removal efficiencies were seen with higher HLR and shorter HRT. Similar removal efficiencies for nitrogen and phosphorus have been seen at higher HLR. This would also increase the efficiency of the system as it would allow treatment of larger quantities of water in a smaller time frame (Fountoulakis *et al.*, 2009).

2.10 Conclusion

The efficiency of biofiltration systems is strongly influenced by HLR and HRT. These systems should not be overloaded and should be given sufficient retention time to ensure that treatment potential is reached (Perales-Momparler *et al.*, 2014). These systems are also economically viable and combat issues such as flooding that plague slum settlements (Parkinson *et al.*, 2007). There is potential to implement biofiltration methods in slums. These methods could potentially clean the polluted water that is currently being untreated and improve the living conditions of people who are living in squalor. Slums are usually located in low lying areas that are usually more prone to flooding (Parkinson *et al.*, 2007). Slums also lack proper sanitary drainage facilities, and this creates unhealthy conditions for slum dwellers. Biofiltration provide an economically feasible opportunity to improve sanitary conditions for slum dwellers.

Chapter 3. Research Methods

To reiterate the aim of this research is to test the capabilities of biofiltration cells to treat contaminated surface water from an informal settlement. This study will specifically examine the hydraulic retention time and hydraulic loading rate as the two main variables that test efficiencies in treating water. Two research questions that were posed for this study.

- Can the treated effluent be safely discharged into freshwater?
- Has the water been sufficiently treated so that it can be repurposed for human activities?

The research design involves collecting water quality data from six biofiltration cells and the inflowing water that is abstracted directly from the Stiebeuel River. The capacity of each biofiltration cell without the packed natural material is 420 m³. Three of the cells were planted with three species of reeds and the others were left unplanted. Samples were collected each week from each individual cell over a period of 12 weeks. The field study involved sampling and analysing water quality of effluent from each biofiltration cell which was then compared for HLRs and HRTs of the respective cells to determine the filtration capability of the cells.

3.1 Design Layout of the biofiltration cells

The influent to each cell is controlled by a network of pipes and flow into each cell is controlled by a ball valve (Figure 6). Each cell is lined with a thick, durable plastic sheet to combat leakages and wear and tear. Compacted fill was applied at the bottom of each cell to create an even surface before the plastic sheet was inserted into the cell. Once complete, the filter media of large stones, small stones and peach pips was placed in the respective cells.

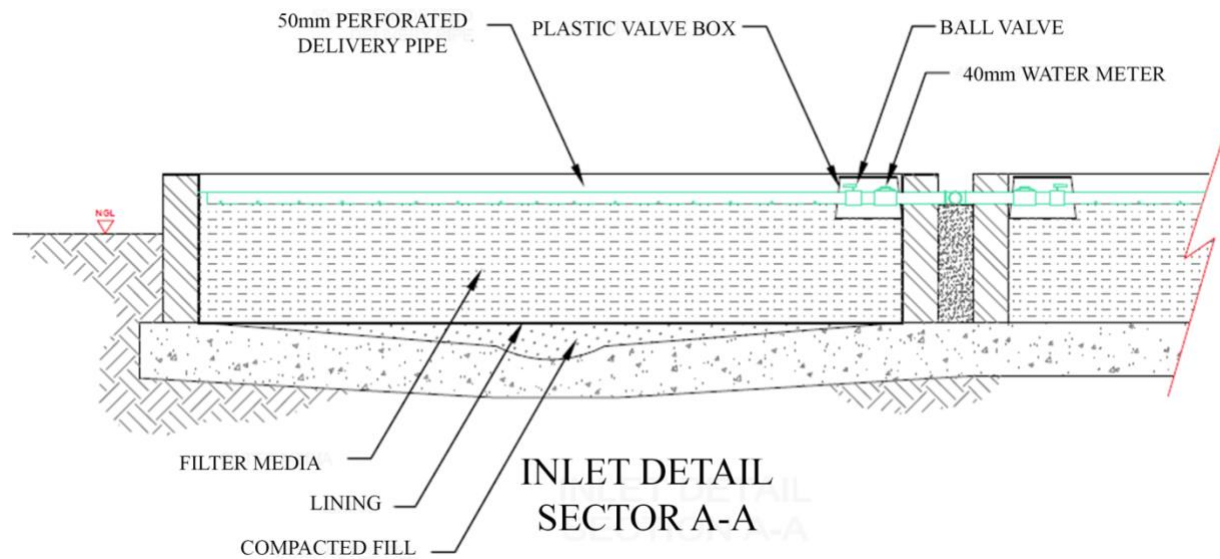


Figure 6: Inlet Detail

Treated water was discharged from each cell as shown in Figure 7. The U-bend in the outlet pipe ensured that the volume of water in each cell remained just 3 to 5 cm from the surface of the cell medium. Water could be released by removing the U-bend coupling.

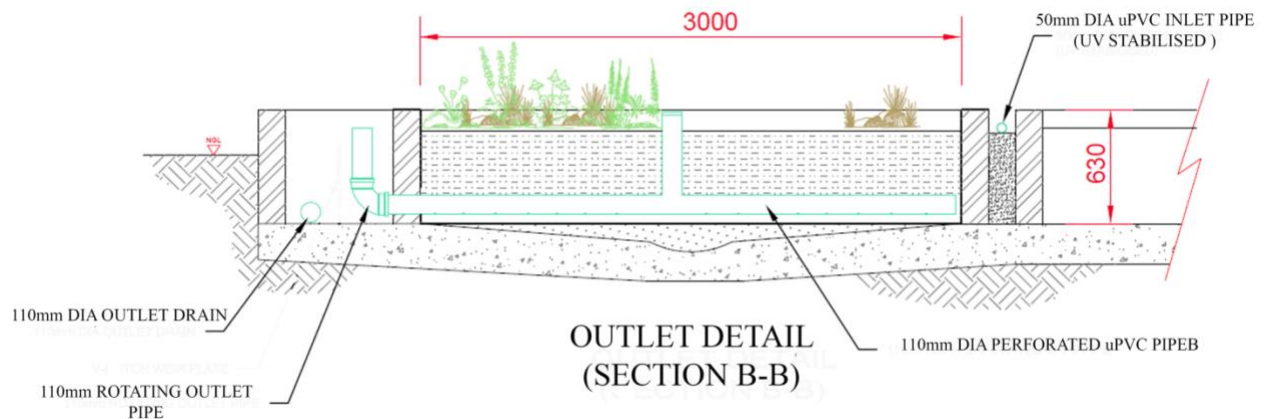


Figure 7: Outlet Detail

3.2 Method

The Water Hub site is the site of an old waste water treatment plant that was built in the 1960s. Once it was abandoned in 2013, some of the infrastructure was vandalised or found in a ruinous state due to aging infrastructure. Drying beds for draining the solid precipitate. It made sense to convert these drying beds into biofiltration cells. The initial appearance of the biofiltration cells before undergoing renovation in 2016 is shown in the photograph in Figure 8.



Figure 8: The state of the drying beds before rehabilitation

The drying beds were excavated and all the material and vegetation from the beds were removed and safely disposed. The walls and plaster of the cells were repaired and outlet catch pits were constructed at the downstream end of each cell.

An assortment of large stones (35 to 50 cm diameter) were placed at the inlet and outlet ends of each biofiltration cell. The rocks protect the perforated drainage pipes and enable water to be distributed evenly at the inlet end of the cell. An inlet valve controls flow which is housed in box with the raised green lid shown in Figure 9. A plastic container is a connection point that allows the upstream cell to be connected for future experiments which will take into account the potential to use multiple cells in tandem to filter the water.



Figure 9: Large rocks placed in the inlet and outlet of two separate biofiltration cells

Multiple inspection pipes were added in each drying bed as shown in Figure 10. These pipes could be used as sampling points at various stages in the drying bed although this option was not used in this study which focused only on the quality of water at the final discharge point of each cell.



Figure 10: Perforated inspection pipes

This study tested three vegetated biofiltration cells and three non-vegetated biofiltration cells. Indigenous plants were acquired from a mature, overgrown wetland in the lower section of the Liesbeek River in Cape Town and transported to the site.

Once the construction of biofiltration cells was complete, a 25 mm pipes was connected to a small submersible pump placed in the Stiebeuel River alongside. Water was pumped to two 10,000 litre tanks and then into the network of pipelines alongside each of cells. As explained earlier, the flow into each cell was controlled by an inlet valve.

3.3 Hydraulic Loading Rate

The hydraulic loading rate was calculated using a flow meter and an analogue counter to determine the volume in cubic metres. The volume of water between each sampling interval was recorded and used to determine the HLR.

3.4 Flushing regimes

The biofiltration cells were flushed for the first 8 weeks of testing. This entailed emptying out each cell at the end of the test period. Initially the retention period was confined to 7 days but this varied later in the experiment in order to assess the experience of other researchers who found that a longer retention time improved the overall treatment. This same retention and flushing procedure was repeated for each cell throughout the study. The cells were not flushed from weeks 9 to 12 but the same HLR as the case for the first 8 weeks. When the cells overflowed, the discharge was directed into to a vegetated bioswale 40 m in length which acted as a conduit before releasing the water back to the Stiebeuel River.

3.5 Vegetation selected for planted cells

Readily available indigenous reeds were selected for the vegetated sections of the biofiltration plants. There species were used in a greenhouse study by the University of Cape Town and were known for their ability to adsorb nutrients from vertically irrigated test beds (Milandri *et al.*, 2012).

3.5.1 Phragmites Australis

The most common species used for vegetation in wetlands systems throughout the world is *phragmites australis* (*P. australis*). *P. australis* is also known as the common reed (Lee & Scholz, 2007). It is well established that some macrophytes such as *P. australis* produce root secretions are toxic to a variety of harmful bacteria such as *E. coli*. *P. australis* is a cosmopolitan species that displayed the ability to reduce the environmental contamination of its surroundings. The plant is able to survive in extreme conditions such as varying temperatures that include extremely warm and extremely cold habitats. It is also able to adapt to wet and dry conditions and to survive a variety of environmental conditions (Srivastava *et al.*, 2014). The global distribution of *P. australis* shows the plant's competitive nature and to adapt to different conditions. It can grow on a wide range of soils with different salinity, fertility, pH and can continue to attain high levels of productivity (Dinka & Szeglet, 1998). A combination of its ability to adapt to a wide range of environmental conditions and phytoremediation potential makes this an appropriate plant for treating contaminated water according to Srivastava (2014).

According to Decamp (2000) constructed wetlands that are vegetated with *P. australis* have the potential to contribute to the removal coliform bacteria by more than 99%. The common reed has also shown the ability to remove faecal streptococci by more than 98%. The common reed biofiltration cells should have a positive effect in filtering polluted water in both monoculture or polyculture forms. *P. australis* has exhibited the ability to remove nutrients from contaminated water. The plant has the potential to remove nitrogen from contaminated water where bacteria and microbial organisms are active around the root zone of the plant (Ah Lee *et al.*, 2006).

3.5.2 Typha Capensis

T. capensis is also known as cattail or bulrush and is an aquatic plant that has potential in treating polluted water. *T. capensis* can grow extremely quickly in aquatic environments such as streams, ponds and marshes. Its distribution can be found throughout southern Africa (Goldblatt & Manning, 2000). The plant tolerates various pH levels and salinity. *T. capensis* can accumulate large amounts of heavy metals and nutrients and store contaminants in the roots (Ma, 2005) A

study by Mudavanhu (2014) has shown that *T. capensis* can treat wastewater with high levels of TDS (total dissolved solids) and EC (electrical conductivity).

Since, *T. capensis* can accumulate nutrients and heavy metals it can be used as a bio- monitor and as an additional form of treatment for polluted water. According to Ma (2005) *T. capensis* was more effective at removing phosphorous than *P. australis*. Ma (2005) mentions further that *T. capensis* tissues can accumulate more zinc, iron and manganese than *P. australis*. Accordingly, *T. capensis* would be a more suitable biological monitor for those metals than *P. australis*. However, *P. australis* has exhibited greater adsorption of lead and therefore would be a better monitor for lead contamination. Both plants displayed similar levels of accumulation for copper and cadmium. Therefore, *T. capensis* has high potential in treating water contaminated with heavy metals. According to a study by Milandri (2012) *T. capensis* performed better than the control used for treating stormwater. The results showcased 86% removal of phosphate, 93% reduction of ammonia and a 56% reduction in nitrates. This was higher than the control in all cases and thus demonstrates that *T. capensis* has remediation capabilities and can provide additional water treatment along with the substrate.

3.5.3 *Cyperus textilis*

C. textilis also known as umbrella sedge was applied in the biofiltration cell with the small stones substrate. The umbrella sedge is native to this region and can be found from the Western Cape to southern KwaZulu-Natal (Goldblatt & Manning, 2000). It is known to grow along streams, river banks, brackish estuaries and coastal wetlands. This species is robust, versatile and produces a large quantity of biomass. *C. textilis* can grow rapidly and has an extensive root system that can grow in various ecological conditions (Goldblatt & Manning, 2000). The umbrella sedge is usually found in inundated areas and thus can tolerate flood conditions. The plant has also been found to burgeon in well drained soils (Van Wyk *et al.*, 2000). However, *C. textilis* can also tolerate a low water table after the community has established itself. The communities consist of dense clumps that average a height of 1.4 m. *C. textilis* is associated with low levels of nitrates and thus should reduce nitrate levels of polluted water (Van Wyk *et al.*, 2000). Reed beds such as *C. textilis* are used throughout the world to clean industrial effluents and contaminated water. The umbrella sedge absorbs excess nitrogen and phosphorous from treated sewage. The plant has also shown to

reduce phenolic compounds and heavy metals from wastewater (Goldblatt & Manning, 2000). According to a study by Van Wyk (2000), *C. textilis* can retain heavy metals such as aluminium and iron. The retention mostly occurred in the root systems, although some retention does occur in the shoot system.

The modern usage of sedges involves growing them in artificially constructed water purification beds such as biofiltration cells. The rhizomes in the sedges can grow without oxygen for a limited time. These plants can also be used for horticulture purposes (Goldblatt & Manning, 2000). *C. textilis* has been used by people in rural areas for crafting and handiworks as well. This can provide a new economy for poorer regions and provide a livelihood for local populations. In the Khanyayo village located in the Eastern Cape, the Mpondo people have direct uses for umbrella sedges. They create mats, baskets and other handicraft items. In fact, there have been cases of overharvesting sedges due to their demand among the Mpondo people (Kepe, 2003). Thus, *C. textilis* has tremendous potential for providing economic incentives to the citizens of the area apart from cleaning contaminated water.

3.6 Substrate

According to Vymazal (2005) the most frequently used media for filtration purposes are crushed rock and gravel. There are several outlet and inlet designs. Normally a rock trench was present at the end of the drying bed. This was implemented to sustain uniform flow across the depth and width of the drying bed (Vymazal *et al.*, 2006). The effluent withdrawal can be a perforated pipe or a single effluent pipe in the rock filled channel (Conley *et al.*, 1991). A single effluent pipe was used for this study. The bottom of each drying bed should be sealed and covered with waterproof membrane as this prevents leaching (Masi & Martinuzzi, 2010). The substrate in the root zone of the drying bed was directly responsible for removing pollutants via chemical and physical processes. The direct removal of pollutant processes included filtration of pathogens and suspended solids in conjunction with sorption and precipitation of pollutants that were biologically degraded via microorganisms. Substrates also provided a base for plant growth and provide additional microbial growth (Conley, *et al.*, 1991).

Microbial activities have been recognized to be a major process in removing nutrients according to Korboulewsky (2012). This is especially the case with nitrogen, 81% of Nitrogen is typically removed via nitrification-denitrification processes (Korboulewsky *et al.*, 2012). According to Wolverton (1986) assimilating macrophytes with microbe rock filters have produced promising wastewater treatment results. Macrophytes and rock filtration assist in establishing biological processes in the drying beds. Once microbial activity is initiated, the microbes establish themselves on the roots of the plant and the rock filters. This develops a symbiosis in the biofiltration cell and improves wastewater treatment ability. Newer technologies try to ensure that there is a large surface area for the rock filters in coexistence with the macrophytes. Increased concentration of microorganisms has been associated with the plant and rock surfaces (Wolverton, 1986).

While rock filters have shown tremendous remediation potential, apricot stones have also shown that they are effective in treating heavy metals from polluted water according (Kazemipour *et al.*, 2008). Kazemipour (2008) used a formula to calculate the heavy metal removal efficiency of apricot pits. Apricot pips removed 60% of Zinc, 95.5% of Copper, 89.6% of Lead and 86% of cadmium and thus have potential in removing heavy metals from polluted water. Apricot pips are also a high-quality raw material that can develop activated carbons. The activated carbon that is produced from the pips have high surface areas which improves microbial activity. The activated carbon has adsorption capacity for Pb, Cd, Zn and Cu. The cost of the adsorbents should be low compared to other filtration materials as apricot pips should be available in larger quantities. Peach pips come under the same genus “*prunus*” as apricot pips and thus could potentially have similar remediation capabilities as apricot pips (Kurz *et al.*, 2008).

3.7 Data Analysis

As mentioned earlier, water samples were collected from each week for a period of 12 weeks from each biofiltration cells and the influent. Measurements and tests were conducted to determine pH, DO, temperature and EC using Ohaus waterproof pen meters.

The following Ohaus models of pen meters were used : pH pen ST20; DO and temperature: DO pen ST 20 D; and EC pen ST20 GB

These results were measured during each site visit at the point of sample collection. However, microbiological and chemical samples were packed in a cooler bag and transported to a water laboratory at the University of Cape Town where tests for a variety of nutrients while *E. coli* sent to Bemlab laboratories. The following methods and practices were used:

Table 1: Water quality criterion tested

Parameter tested	Method
Ammonia	The salicylate method for powder methods was used to test for ammonia from each of the collected water samples. A Hach DR 2700 spectrometer was used to calibrate and test each sample. (Water Analysis Handbook).
Nitrate	The cadmium reduction method for powder pillows was used to test for nitrates from each water sample. The Hach DR 2700 had a stored program in place for testing nitrates. (Water Analysis Handbook).
Nitrite	The diazotization method for powder pillows was utilized to determine the nitrite content from each water sample. The Hach DR 2700 used a stored program in place to test for nitrites. (Water Analysis Handbook).
Orthophosphate	The ascorbic acid method for powder pillows method was used to discover the orthophosphate levels of each sample from the Water Hub. A Hach DR 2700 spectrometer was used to calibrate and test each sample. (Water Analysis Handbook).

3.8 Limitations

One of the key limitations for this study was the lack of control over environmental conditions. The biofiltration cells were not covered and therefore rainfall could have diluted the water in the cells. This additional dilution could have improved water quality. This improvement would have been attributed to the cells.

Another key limitation for this study was temperature. Low temperatures can negatively impact wetland treatment efficacy (Kadlec & Reddy, 2001). Thus, further analysis should be done during warmer parts of the year to determine whether treatment efficiencies have improved.

Lastly, the state of contaminated water varied greatly throughout the study. It would have been advantageous to provide an influent source that displayed consistent water quality parameters. This would have provided greater clarity on each cell's treatment efficiency.

Chapter 4. Results and Discussion

4.1 Introduction

Water quality data were collected each week in August to October 2017 and uses pH, electrical conductivity (EC), Dissolved Oxygen (D.O.), Ammonia (NH₃), Orthophosphate (PO₄³⁻) and Nitrates (NO₃⁻) as pre-and post-treatment indicators of contamination. Each parameter was tested and compared against the South African Water Quality Guidelines (DWAF, 1996). These guidelines assisted in understanding how the water conformed to standards for reuse. F-values and p-values were calculated by comparing influent values with each biofiltration cell. Bacteria tests were conducted on three separate occasions. As mentioned earlier, each of the six biofiltration cells contain different media, namely large stone aggregate (19 to 25 mm), smaller stone aggregate (7 to 9 mm), and peach pips (or stones) as a carbon source. The experiment aimed at to determine which of the six cells performed best in cleaning the influent water, and in determining the detention time and its influence on the resultant water quality. The analysis begins with a description of the data and a decision on an appropriate statistical model to determine the significance of the post-treatment samples compared to the influent. For the sake of brevity, the presentation of the results and analysis is confined to the best performing cells. Data on the performance of the other cells are presented in the Appendices.

Raw data were ‘cleaned’ to remove negative values and left blank and other capturing errors that were obvious or were extreme outliers were removed. This was done to standardise the data and remove values that may conflate the results. Table 2 below shows the code names of each biofiltration cell and of the influent.

Table 2: Code-names for six biofiltration cells and the influent

INF	Influent
LSV	Large stones with vegetation
LS	Large stones
SSV	Small stones with vegetation
SS	Small stones
PP	Peach pips
PPV	Peach pips with vegetation

Data Analysis and Results

This section describes the results and data analysis of each cell for parameters mentioned above. Data were analysed using the statistical package ‘SPSS’.

4.2 First Stage- Testing for Normality

A test for normality was conducted to determine the data distribution and hence the kind of analytical tests that would be suitable to calculate the difference between the influent and the six

samples. The test for normality was done by calculating the descriptive statistics for each sample as well as the influent.

Most of the sample data was not normally distributed. It exhibited a skewness value greater than 1 and kurtosis values larger than 3. This is illustrated in the descriptive tables below as well as in the box and whisker plot figures which show the boxplots of the data. The skewness, small sample size and large standard deviation meant using non-parametric tests to explain the difference between the means of various parameters.

4.3 pH results for Biofiltration cells

Table 3: Descriptive statistics for pH samples

	N	Min	Max	Mean	Std. Dev	Variance	Skewness		Kurtosis	
							Statistic	Std. Error	Statistic	Std. Error
INF	12	6.98	7.48	7.2317	.16425	.027	.404	.637	-.926	1.232
LSV	12	6.29	7.35	7.0192	.27936	.078	-1.633 ^b	.637	3.852 ^a	1.232
LS	12	6.22	6.98	6.6917	.20854	.043	-.939	.637	1.436	1.232
PPV	12	5.00	6.86	5.9267	.60519	.366	.155	.637	-.773	1.232
PP	12	4.91	6.70	5.7325	.49940	.249	.184	.637	.123	1.232
SSV	11	6.23	6.71	6.4050	.13481	.018	1.005 ^b	.637	1.145	1.232
SS	11	5.75	6.54	6.2692	.21172	.045	-1.320 ^b	.637	2.714	1.232

^aKurtosis for LSV <3

^bSkewness >-1 or <1

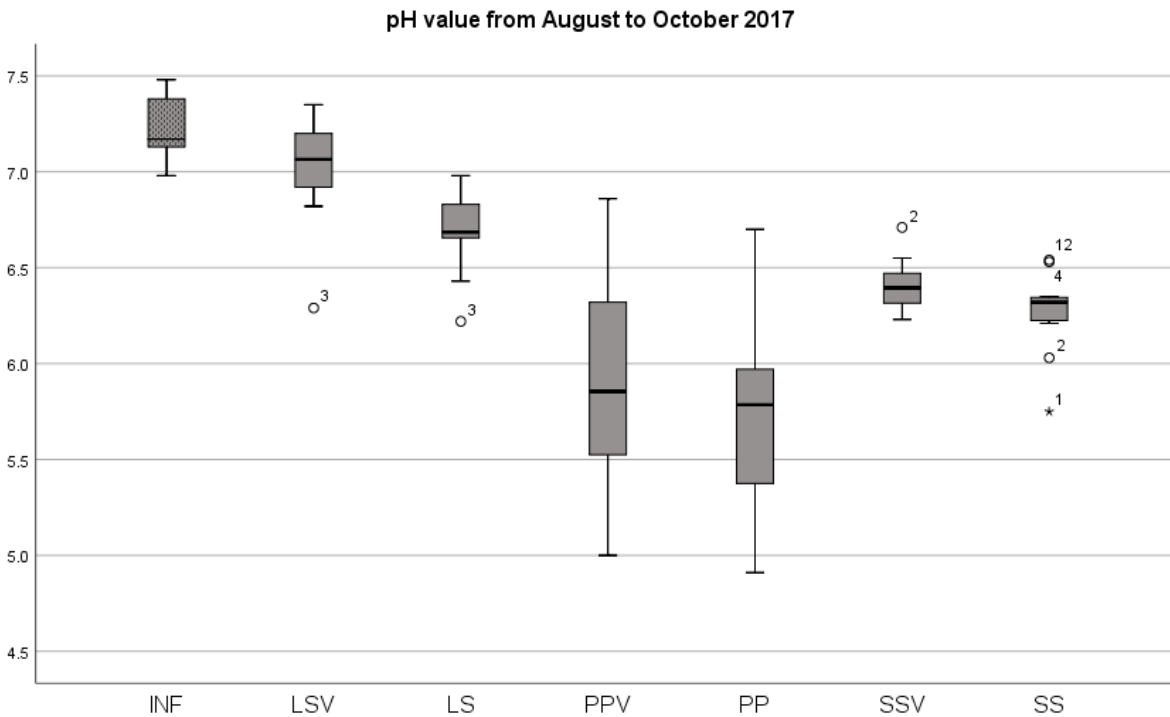


Figure 11: Box and Whisker plot of pH

Table 4: pH parameters for irrigation (DWAF, 1996)

pH Range	Crop Yield and Quality	Sustainability	Irrigation Equipment
< 6.5	Increasing problems with foliar damage when crop foliage is wet. This could give rise to yield reduction or a decrease in the quality of marketable materials	Increasing problems with the availability of several micro- and macro-nutrients in toxic concentrations are experienced in this range over the long term	Increasing problems with corrosion of metal and concrete in irrigation equipment are experienced in this range
Target Water Quality Range 6.5 - 8.4	Even when crop foliage is wetted, this should not cause foliar damage in plants which will result in a yield reduction in the quality of marketable products.	Soil pH within this range does not present major problems with either unavailability of plant nutrients or toxic levels of elements.	Mostly no major problem with either corrosion irrigation equipment is experienced. Slight to moderate problems with the clogging of drip irrigation systems.

As shown by Table 4, pH levels below 6.5 can cause foliar damage and increase complications with the availability of nutrients that can be toxic long term. These levels can cause corrosion of irrigation equipment and thus increase irrigation expenses unless drip irrigation systems are utilized. However, if pH levels are kept within the range of 6.5-8.4, then the pH guidelines for irrigation are met (DWAF, 1996). Figure 11 shows the pH range for every biofiltration cell.

However, pH values in LSV and LS cells became more acidic throughout the sampling process. There were only two occasions where a pH increase occurred as displayed by Table 5. The pH increases were only seen in the LSV cells and happened during the first and fourth sampling points. HLR and HRT did not affect pH. Sample points including both low and high HLR resulted in significant differences in pH. Longer HRT also did not affect pH values. Large differences are seen at both high and low HLR values.

Table 5: pH differences for influent, LSV and LS

Sample	Influent	LSV	Difference	HLR (m ₃)	LS	Difference	HLR (m ₃)	HRT (days)
1	7.17	7.28	-0.11		6.98	0.19		
2	7.45	6.92	0.53	2781	6.68	0.77	2510	7
3	7.17	6.29	0.88	2,199	6.22	0.95	1171	7
4	6.98	7.26	-0.28	2,069	6.69	0.29	3596	7
5	7.07	7.07	0	1,462	6.64	0.43	2191	7
6	7.16	7.14	0.02	2,358	6.72	0.44	4198	14
7	7.11	6.92	0.19	1,974	6.93	0.18	2243	7
8	7.31	6.82	0.49	4,028	6.43	0.88	5142	7
9	7.26	6.99	0.27	1,955	6.88	0.38	2598	7
10	7.47	7.35	0.12	2,559	6.68	0.79	2798	14
11	7.15	7.06	0.09	4,283	6.67	0.48	4431	21
12	7.48	7.13	0.35	1,255	6.78	0.7	2029	28

Table 4 presents the target pH values for irrigation use should be between 6.5 and 8.4. LSV maintained these values throughout the study except on one occasion where a pH of 6.29 was observed (see Table 5). However, LSV on average decreased pH values throughout the study. If the influent was acidic (pH under 7), then LSV may not have met pH standards for irrigation use. LS also decreased pH values throughout the study. There were two occasions where requirements for irrigation standards were not met. As displayed in Table 4, utilizing water with pH levels below 6.5 could reduce crop yields and negatively impact conventional irrigation systems due to the foliar damage that can occur (DWAF, 1996). Varying HLR and HRT did not affect pH values. LSV met irrigation standards more often than any other cell in this study.

4.4 E.C results for Biofiltration cells

Table 6: Descriptive statistics for EC samples

	N	Min	Ma x	Mean	Std. Dev	Variance	Skewness		Kurtosis	
INF	12	188	347	247.00	45.806	2098.182	.677	.637	.624	1.232
LSV	12	200	334	262.25	33.117	1096.750	.198	.637	1.835	1.232
LS	12	174	255	212.58	26.919	724.629	- .086	.637	-1.124	1.232
PPV	12	113	274	195.50	52.663	2773.364	.007	.637	-1.565	1.232
PP	12	159	280	196.08	34.156	1166.629	1.29 3	.637	2.317	1.232
SSV	11	190	319	226.45	44.478	1978.273	1.35 4	.661	.612	1.279
SS	11	152	283	194.36	40.739	1659.655	1.58 5	.661	1.633	1.279

Table 7: E.C Parameters (Bauder *et al.*, 2011)

E.C level	Salinity hazard
$\leq 750 \mu\text{S}$	None
750- 1,500 μS	Some

As shown by Table 6, LSV was the only cell that had a higher mean than the influent. However, the influent and all the biofiltration cells met the irrigation standards given in Table 7. E.C levels that are between 750 μS and 1,500 μS can pose a salinity risk for crop production. Increased salinity leads to lower producing crop yields (Machado & Serralheiro, 2017). For this study, HRT and HLR did not influence E.C levels for LSV, LS, PPV and PP. Figure 12 shows the different ranges for E.C for each biofiltration cell.

The largest decreases in E.C for SSV and SS cells was 153 μS and 163 μS respectively. These reductions were seen with the high loading rates of 3511 m^3 and 3819 m^3 for SSV and SS respectively. These were the highest HLR applied to these cells during the study. Thus, high loading rates could potentially lead to decreases in EC for SSV and SS. Increasing HRT did not to influence EC values for SSV and SS.

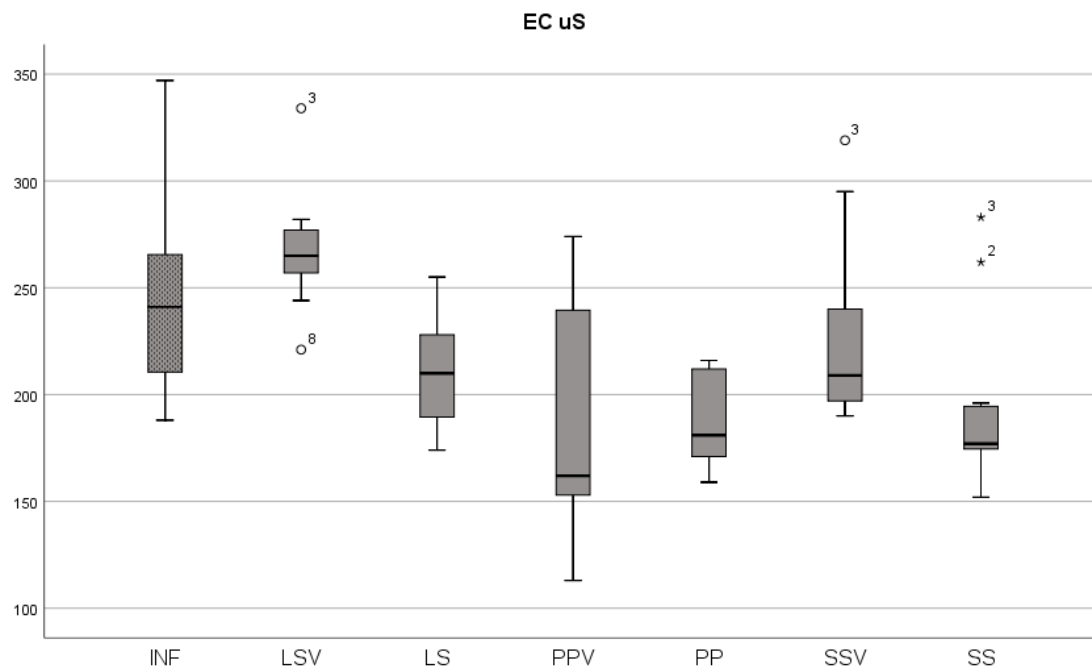


Figure 12: Box and Whisker plot of E.C

4.5 DO results for Biofiltration cells

Table 8: Descriptive statistics for Dissolved Oxygen samples

	N	Min	Max	Mean	Std. Dev	Variance	Skewness		Kurtosis	
INF	1 2	1.4	5.4	2.808	1.2652	1.601	.801	.637	- .040	1.232
LS V	1 2	1.3	2.7	1.950	.3729	.139	.208	.637	.534	1.232
LS	1 2	1.0	3.1	1.583	.6965	.485	1.666	.637	1.72 5	1.232
PPV	1 2	.0	1.7	1.133	.4619	.213	-1.312	.637	2.37 9	1.232
PP	1 2	.1	7.0	1.175	1.8912	3.577	3.130	.637	10.1 94	1.232
SSV	1 1	1.1	3.8	2.200	.9400	.884	.452	.637	- 1.03 1	1.232
SS	1 1	1.0	4.4	1.900	.934	.871	1.897	.637	4.40 2	1.232

Table 9: DO parameters (Behar *et al.*, 1996)

Dissolved Oxygen Levels	Water Quality Requirements
0-2 mg/L	not enough oxygen to support life
2-4 mg/L:	only a few fish and aquatic insects can survive.
4-7 mg/L	good for many aquatic animals, low for cold water fish
7-11 mg/L	very good for most stream fish

Table 9 displays how DO levels can affect aquatic life in freshwater ecosystems. Dissolved oxygen involves the oxygen mixed in water and made available to aquatic species for respiration. It is a critical component required to sustain life in aquatic ecosystems (Behar *et al.*, 1996). As shown by Table 8, the highest DO level seen for any cell was 4.4 mg/l that occurred for SS cell.

Both LSV and LS created an anaerobic environment in the cells and lowered DO levels. Thus, they would not have met the standards set by Table 9. Only SSV had a mean above 2 mg/l, which would only meet minimal requirements for healthy ecosystems. However, even with low DO levels, mosquito larvae were found in PP. SSV and SS performed the best among the biofiltration cells. However, DO levels were still low for both cells and would only support resilient aquatic species. The influent had a larger mean than any cell. Overall, the biofiltration cells reduced DO levels as seen by Figure 13. HLR and HRT did not appear to influence DO levels for any cell.

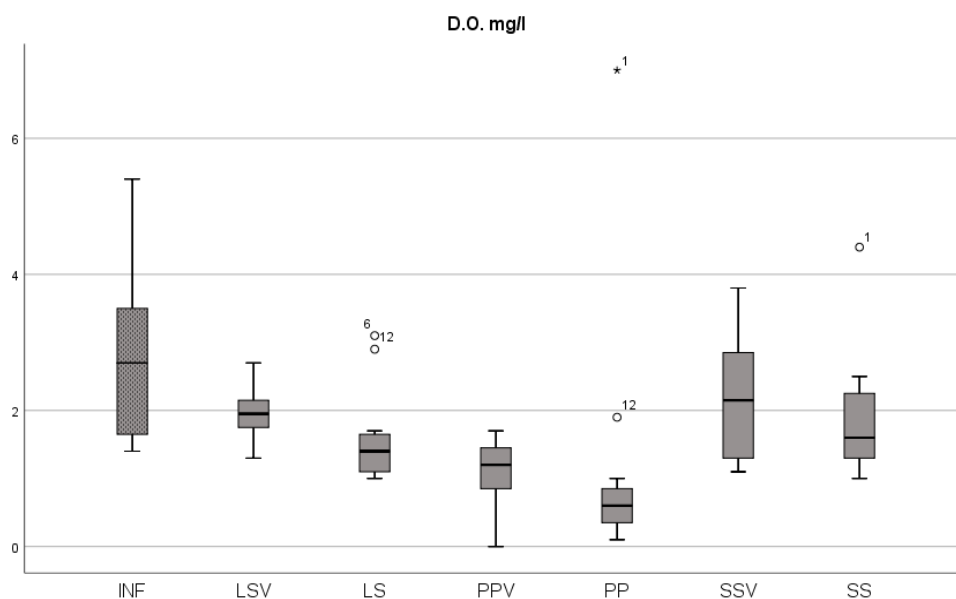


Figure 13: Box and Whisker plot of DO

4.5 Ammonia (NH₃) results for Biofiltration cells

Table 10: Descriptive statistics for Ammonia (NH₃) samples

	N	Min	Max	Mean	Std. Dev	Variance	Skewness		Kurtosis	
							Statistic	Std. Error	Statistic	Std. Error
INF	12	2.4	13.5	5.7	2.7	7.759	2.070	.637	5.842	1.232
LSV	12	.06	7.24	2.37	2.5	6.462	.885	.637	-.514	1.232
LS	12	.1	7.75	2.82	2.3	5.381	.679	.637	.386	1.232
PPV	12	1.3	9.25	5.19	2.5	6.270	.141	.637	-1.044	1.232
PP	12	2.2	11.25	5.9	2.44	5.973	.532	.637	1.108	1.232
SSV	11	1	6	2.97	1.81	3.282	.689	.661	-.417	1.279
SS	11	0	5	2.86	1.37	1.887	-.697	.661	-.001	1.279

Table 11: Ammonia water quality requirements (World Health Organization, 2003; Office of Environmental Public Health, 2000)

Ammonia levels	Water Quality Requirements
=<0.2 mg/l	Natural levels in groundwater
=< 0.5 mg/l	Drinking Water
=<1.0 mg/l	Toxic to aquatic species

Table 12: Ammonia for drinking purposes (DWAf, 1996)

Ammonia Range (mg/l)	Effects
Target Water Quality Range 0 - 1.0	No health or aesthetic effects
1.0 - 2.0	Possibility of taste and odor complaints from consumers
2.0 - 10.0	Consumer complaints of objectionable taste and odours likely.

Table 11 displays suitable ammonia levels in different water sources. Groundwater ammonia levels should be below 0.2 mg/l. Ammonia levels in drinking water should not exceed 0.5 mg/l. Ammonia levels above 1.0 mg/l are toxic for aquatic species and thus effluents that have ammonia concentrations higher than 1.0 mg/l should not be discharged into freshwater ecosystems (World Health Organization, 2003; Office of Environmental Public Health, 2000). Table 12 describes

ammonia levels suitable for potable use. Ammonia concentrations higher than 2.0 mg/l could possibly receive consumer complaints (DWAF, 1996).

4.5.1 LSV and LS

Both cells on average reduced ammonia concentrations significantly as displayed by Table 10. Figure 14 exhibits how both cells had lower ammonia concentrations than the influent on every occasion. LSV cell observed a reduction of $\geq 90\%$ on four occasions. LS observed $\geq 90\%$ reductions on three occasions. Both cells could significantly reduce ammonia concentrations with high HLR. However, LSV observed reductions of $>96\%$ when HLR was within, 1462 m³-2,358 m³. Increasing HLR in LSV and LS to $>4,000$ m³ led to lower reductions in ammonia. However, the systems still produced significant reductions with higher HLR. On average LSV and LS reduced ammonia concentrations by 65.65% and 56.21%. HRT had a large impact on cell performance. HRT of 7 and 14 days resulted in significant reductions when the cells were flushed. However, longer HRT that was applied from sample points 8-12 resulted in lower reductions as shown in Figure 14. Thus, these systems performed better when the cells were flushed regularly. Longer HRT reduced the systems efficiency. However, concentrations still decreased when longer HRT was used without flushing was applied. An HRT of 28 days led to $>70\%$ decrease in ammonia concentration for both cells.

LSV and LS on average decreased ammonia concentrations as displayed by Table 10. There was never any point of time where ammonia concentrations did not decrease for both cells throughout the duration of the study. However, even though large differences were seen, both cells still had ammonia concentrations higher than 1.0 mg/l, which is toxic to aquatic species as demonstrated by Table 11. LSV reduced ammonia concentrations below 0.2 mg/l on four occasions. LS managed to decrease ammonia below 0.2 mg/l on three occasions. However, apart from those data samples, ammonia concentrations were ≥ 1.0 mg/l on every other occasion and thus would not be suitable for standards set by Table 11. However, ammonia levels were always below 10 mg/l and thus are safe to consume. Influent levels were below 10 mg/l on nearly every occasion and thus would also have met safety standards set by Table 12.

LSV observed significant reductions when HLR ranged from 1,462 m³- 2,358 m³ with an HRT that included 7 and 14 days. LS observed significant reductions when HLR ranged from 2,198 m³- 4,198 m³ with an HRT of 7 and 14 days. LS observed similar reductions with higher HLR than LSV. However, LSV could potentially have seen similar reductions if the same HLR had been applied. During the study LSV performed greater reductions in ammonia concentrations on similar HLR to LS. Both cells performed better when the cells were flushed. When the water was retained and allowed to overflow, HRT was increased to 14, 21 and 28 days. Both cells observed a decrease in ammonia reduction during that period. HRT of 28 day still led to a >70% reduction in ammonia for both cells. However, larger reductions had been seen previously when the cells were flushed. HRT of 7 days was sufficient for large decreases to occur. HLR between 1,900 m³- 4,200 m³ for both cells observed large reductions and similar HLR should be applied and flushed after 7 days.

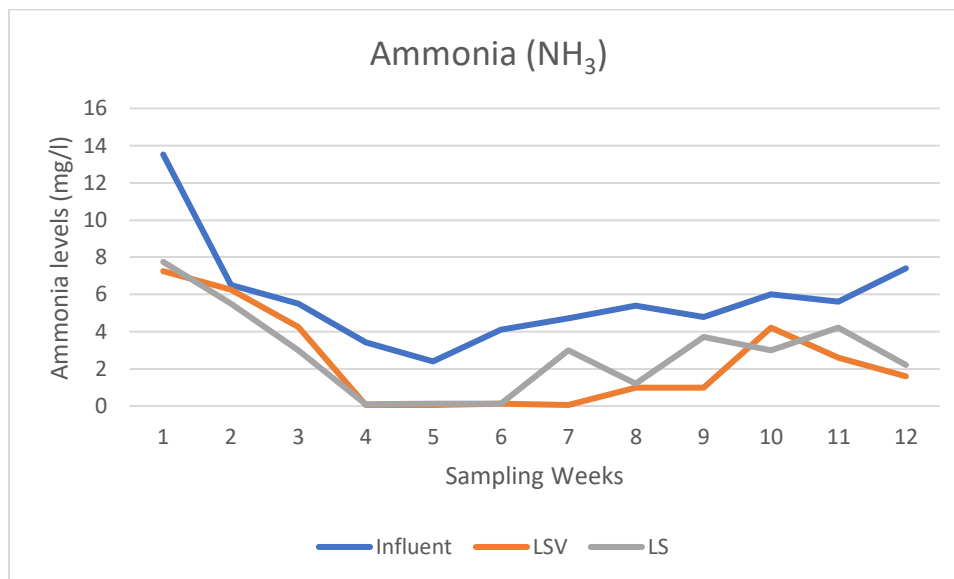


Figure 14: NH₃ comparison between influent, LSV and LS

4.5.2 SSV and SV

SSV and SS decreased ammonia concentrations as shown in Figure 15. Figure 15 displays how SS ammonia levels were consistently below the influent throughout sampling. SSV and SS on average decreased ammonia concentrations by 47.5% and 46.5% respectively. The highest removal efficiency for SSV was 82.98% that occurred with an HLR of 2,606 m³. Lower HLR exhibited lower ammonia reductions. Increasing HRT for SSV led to ammonia reductions. Both HLR and

HRT played a role for SSV. The highest removal efficiency for SS was 95.83% that occurred with an HLR of 3,233 m³. However, large reductions were also seen with low HLR. Both cells lost efficiency when HRT was increased to 21 days. SS cells performed better when the influent was flushed from the cells every week, rather than retained. SSV cells performed well in both conditions, however HRT of 21 days only showed 28.57% reduction. These cells performed better than the peach pip cells but were on average not able to reduce ammonia concentrations as much as LSV and LS as shown in Figure 16.

SSV and SS on average decreased ammonia concentrations for this study as shown in Figure 26. SSV did increase ammonia concentrations on one occasion, however decreases were seen for the rest of the study. SS did not increase ammonia concentrations at any point during this study. SSV decreased ammonia concentrations below 1.0 mg/l on two occasions, while SS achieved it once throughout the study. Thus, as demonstrated by Table 11, effluent ammonia concentrations from these cells would be toxic to aquatic species. However, both cells made the standards set by Table 12.

The largest percentage decrease by SSV and SS was 82.98% and 95.83% respectively. This occurred when HLR for SSV was 2,606 m³ and 3,233 m³ for SS. Both cells fluctuated with HLR. HLR values below 2,000 m³ resulted in lower removals and in the case of SSV increased ammonia concentrations during sample 3 (Figure 26). However, similar loading rates were applied the next week and ammonia concentrations decreased in both cells. Increasing HRT led to ammonia reductions, however when HRT was increased to 21 days, ammonia removal efficiencies decreased in both cells. Thus, HRT of 7-14 days was adequate to decrease ammonia concentrations. SS performance fluctuated regardless of HLR. Throughout the study, SS observed large decreases with an HLR of $\geq 3,000$ m³, however the next week a similar loading rate was applied, and the ammonia reduction was significantly less.

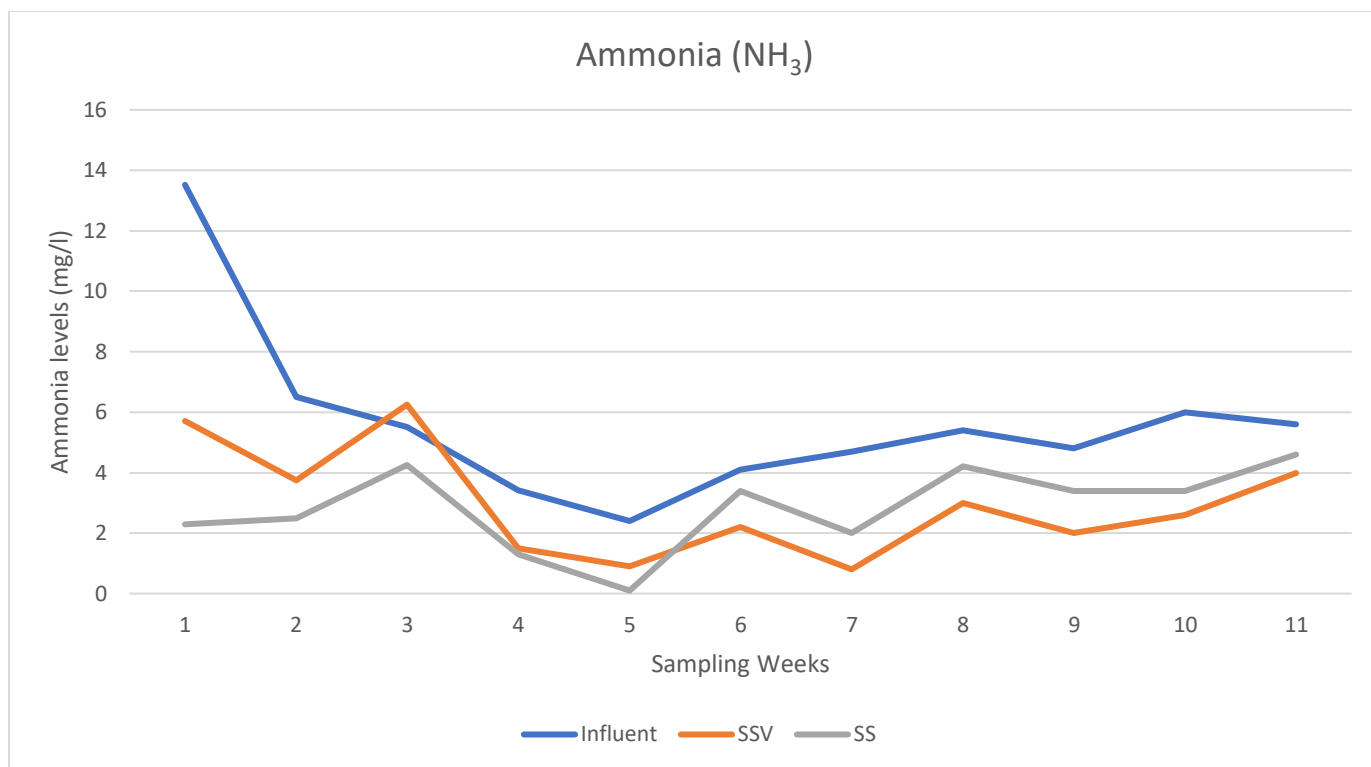


Figure 15: NH₃ comparison between influent, SSV and SS

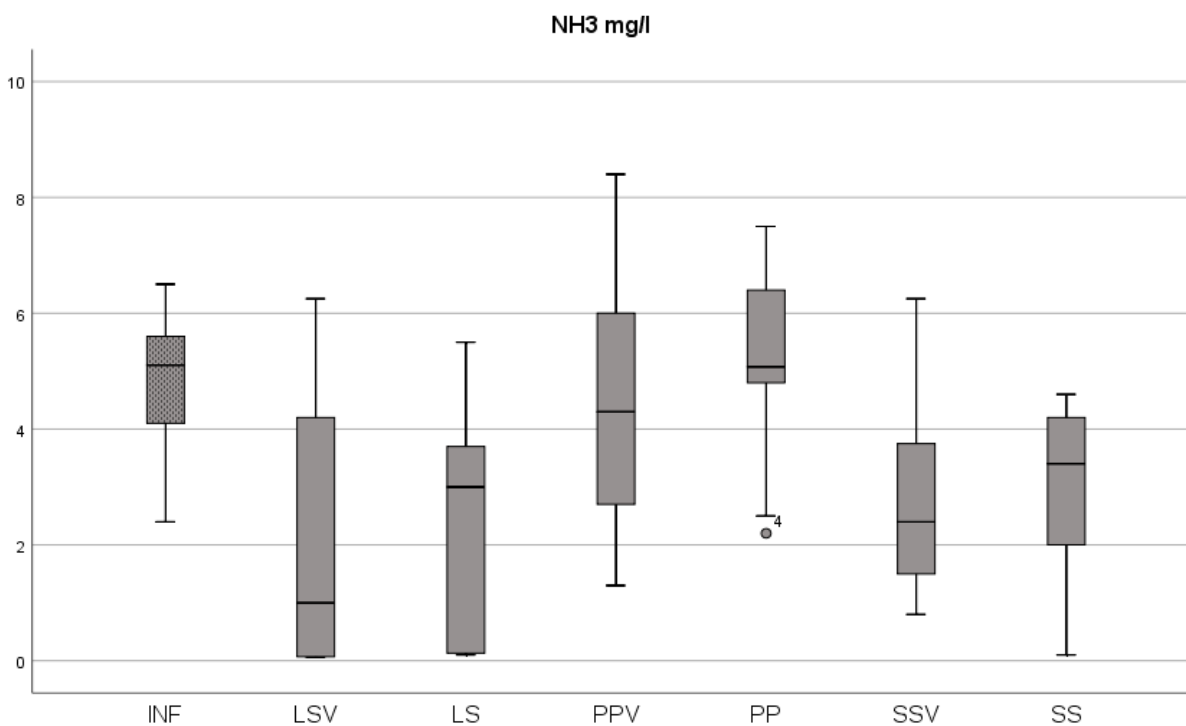


Figure 16: Box and Whisker plot of NH₃

4.6 Orthophosphate results for Biofiltration cells

Table 13: Descriptive statistics for Orthophosphate (PO₄³⁻) samples

	N	Min	Max	Mean	Std. Dev	Variance	Skewness		Kurtosis	
							Statistic	Std. Error	Statistic	Std. Error
INF	12	.42	21.00	4.2742	6.51	42.505	2.193	.637	3.891	1.232
LSV	12	.00	3.40	.6183	.97	.950	2.495	.637	6.613	1.232
LS	12	.19	3.1	.911	.9	.811	1.797	.637	2.527	1.232
PPV	12	.8	22.94	4.35	5.97	35.712	3.211	.637	10.771	1.232
PP	12	.77	19.65	5.07	4.	23.650	2.790	.637	8.867	1.232
SSV	11	0	41	6.66	11.8	139.279	2.873	.661	8.585	1.279
SS	11	0	3	.74	1.006	1.013	1.775	.661	1.995	1.279

Table 14: Orthophosphate parameters (Osmond *et al.*, 1995)

Orthophosphate levels	Disposal
0.1 mg/l	Streams/Rivers
0.05 mg/l	Streams/ Lakes
0.025 mg/l	Lakes/ Reservoirs

Table 14 displays the orthophosphate levels suitable for disposal in streams, rivers, lakes and reservoirs. Orthophosphate level discharge limits for lakes and reservoirs are below 0.025 mg/l. Excessive orthophosphate levels cause eutrophication which degrades the environment (Kundu *et al.*, 2015). Thus, effluent samples should follow orthophosphate guidelines.

These samples were taken during winter time. Orthophosphate removal decreases in wetlands during the winter months. The removal levels could potentially have been higher during the summer time (Chazarenc *et al.*, 2007).

4.6.1 LSV and LS

Both LSV and LS reduced orthophosphate concentrations throughout the study apart from one occasion for LS as displayed by Figure 17. LS increased phosphate levels during sample 7. Figure 19 demonstrates how orthophosphate values were still significantly reduced even when the influent was found to have high levels of orthophosphate. On average LSV and LS reduced orthophosphate by 82.78% and 60.64% respectively. LSV observed 100% reduction on two occasions. LSV observed $\geq 83\%$ on nine occasions. The largest reduction seen in LS was 89.70%. Increasing HRT resulted in high reductions in orthophosphate levels. Both cells observed decreases of $\geq 80\%$ when HLR of $>4,000 \text{ m}^3$ was used. Thus, both cells could reduce orthophosphate concentrations on higher loading rates.

LSV and LS on average reduced orthophosphate levels throughout the study as displayed by Figure 18. LSV displayed 100% orthophosphate removals on two occasions. LS increased contaminant levels on one occasion. However, while both cells showed significant decreases, the resulting effluent would still not be able to meet standards set by Table 14. LSV observed levels $\geq 0.1 \text{ mg/l}$ on three occasions. LS was unable to meet the standards set by Table 14. The influent had high levels of orthophosphates and in the last two weeks of testing observed influent orthophosphate levels of 21 mg/l and 14.7 mg/l . However, both cells could significantly reduce contamination with longer HRT of 21 and 28 days. Both cells observed decreases of 80% when an HLR of $> 4,000 \text{ m}^3$ was used. However, while large decreases were seen, the resulting effluent does not meet standards for disposal into other water bodies. However, the orthophosphate levels could still potentially be adequate for irrigation purposes (Usman, 2013).

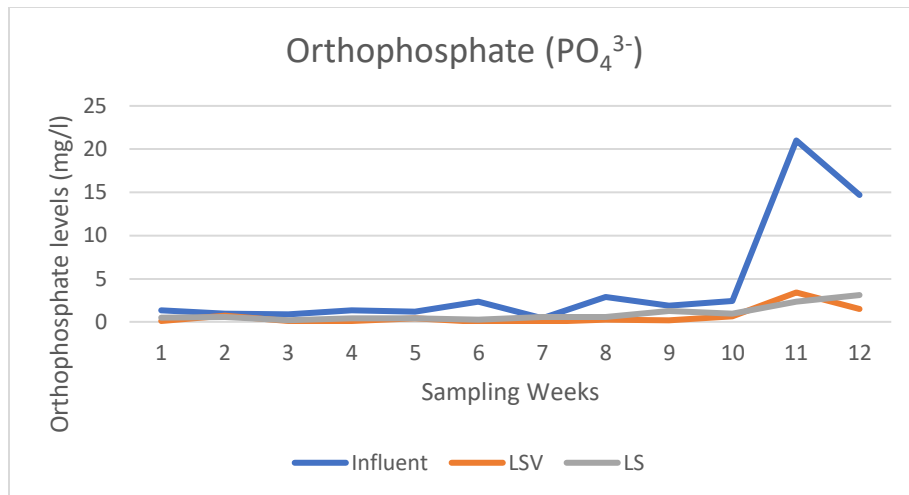


Figure 17: PO₄³⁻-comparison between influent, LSV and LS

4.6.2 SSV and SS

SSV and SS performed conversely throughout the study as shown in Figure 18. SS decreased orthophosphate on every occasion excluding sample 7 as shown in Figure 18. SSV did not perform consistently. Figure 19 demonstrates that SSV had high values of orthophosphate throughout the study. The largest decrease in orthophosphate seen by SSV was 79.05% that occurred with an HRT of 21 days. On averaged SS decreased concentrations by 69.82%. SSV increased concentrations on average by 495.67%. SSV showed an increase of 4,433% during sample 3. This occurred with an HLR of 1,625 m₃, which was one of the lowest loading rates used during the study. SS observed significant reductions with varying loading rates. Increasing HRT to 21 days observed a decrease of 88.57%. SS had a reduction of 98.28% when an HRT of 14 days was used. High HLR \geq 3,000 m₃ appeared to negatively affect removal efficiency. However, when applied with longer HRT, the cells responded well even with high HLR.

SSV and SS performed conversely to each other as shown in Figure 18. SSV increased orthophosphate levels, while SS decreased them. SS observed large decreases throughout the study and would have met the standards set by Table 14 on one occasion. The largest decreases were seen when HRT was 14 and 21 days was applied for SS. HLR \geq 3,000 m₃ did decrease removal efficiency for SS. However, large reductions were still seen when longer HRT was used. SSV observed a large decrease when an HRT of 21 days was applied. SSV observed large increases regardless of HLR. SSV on average performed the most poorly for orthophosphate contamination out of all the biofiltration cells. This could potentially be due to the stimulation of a phosphate

mineralizing enzyme that can be released with re-wetting regimes (Song *et al.*, 2005).

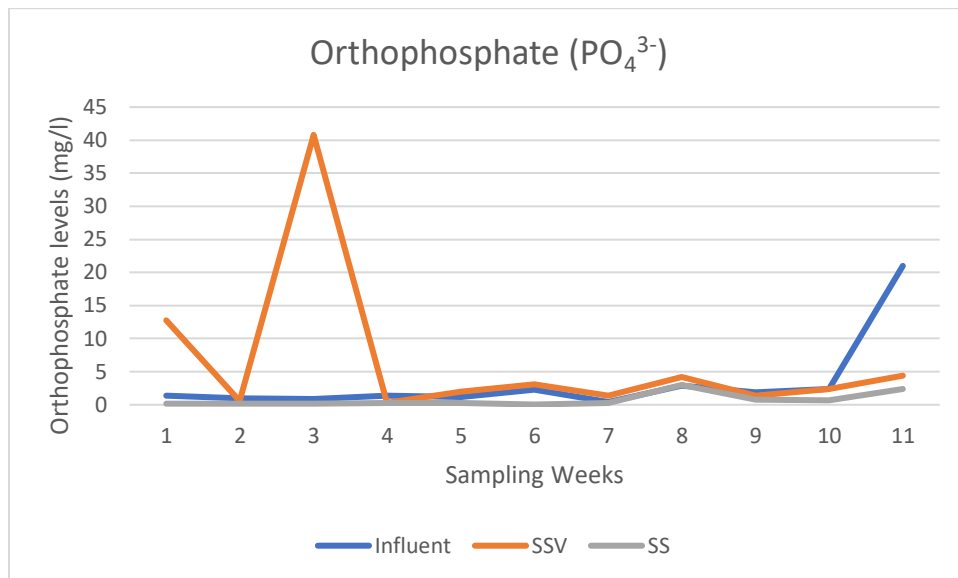


Figure 18: PO_4^{3-} -comparison between influent, SSV and SS

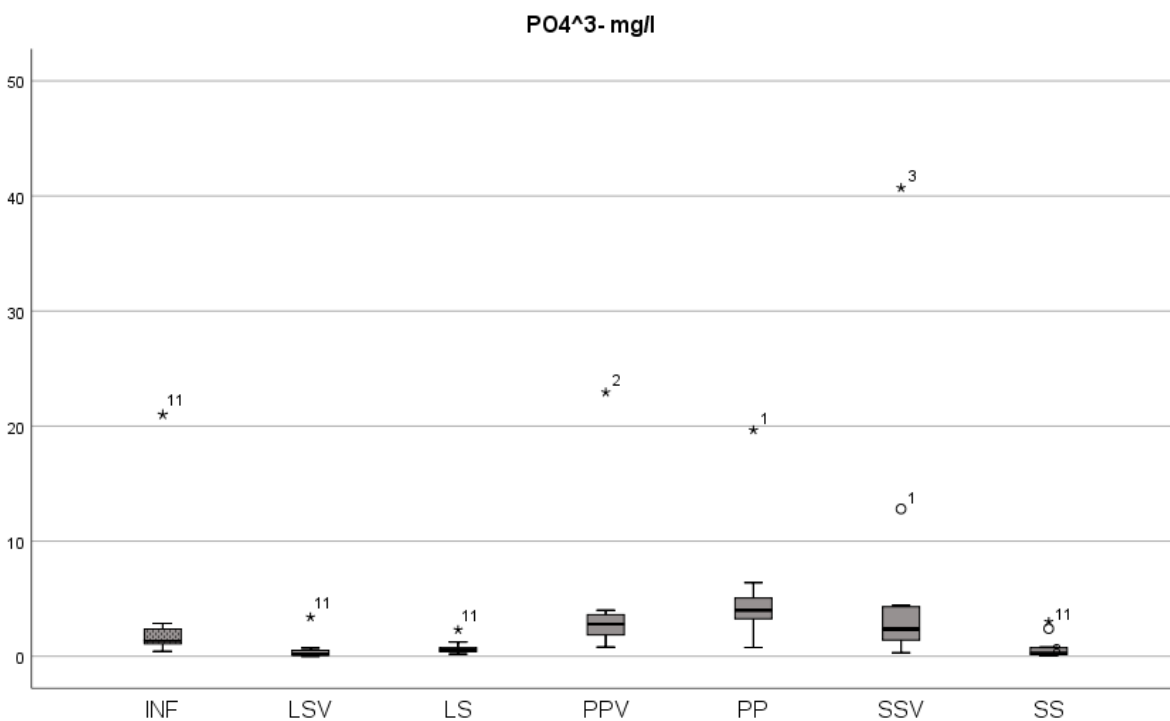


Figure 19: Box and Whisker plot for Orthophosphate

4.7 Nitrates results for Biofiltration cells

Table 15: Descriptive statistics for Nitrates (NO₃-) samples

	N	M in	Max	Mean	Std. Dev	Variance	Skewness		Kurtosis	
							Statistic	Std. Error	Statistic	Std. Error
INF	1 2	.9	2.5	1.633	.4735	.224	.046	.637	-.506	1.232
LSV	1 2	.9	5.2	3.042	1.4557	2.119	.014	.637	-1.367	1.232
LS	1 2	.8	4.6	2.6	1.2	1.510	.253	.637	-1.023	1.232
PPV	1 2	.7	7.9	3.8	2.7	7.795	.466	.637	-1.670	1.232
PP	1 2	.2	7.2	3.41	1.9	3.789	.340	.637	-.154	1.232
SSV	1 1	.0	2.4	1.2	.60	.366	.010	.661	1.747	1.279
SS	1 1	.3	3.8	1.409	1.1004	1.211	1.466	.661	1.238	1.279

Table 16: Nitrate parameters (DWAF, 1996)

Nitrate Range (as mg/l)	Effects
0-6	No adverse health effects
6 - 10	Concentrations in this range generally well tolerated

4.7.1 SSV and SS

SSV and SS were decreasing nitrate concentrations as shown in Table 15. Figure 20 demonstrates how nitrate values increased on two occasions for both SSV and SS. On average, SSV and SS decreased nitrate values by 22.58% and 12.89%. SSV managed to completely remove nitrates by 100%. The highest removal efficiency reached by SS was 72%. SS displayed an increase in nitrates when an HLR of 1,708 m³ was used. This was the second lowest HLR applied and tested for this cell. Increasing HRT to 21 days led to a decrease in nitrate concentrations for both cells. However, greater removal efficiencies had been achieved earlier in the study. SSV and SS were the only cells in this study that on average reduced nitrate concentrations.

SSV and SS on average decreased nitrate concentrations throughout the study as shown in Figure 21. They were the only cells that decreased nitrates levels. Both cells increased nitrate concentrations on only two occasions throughout the study. Increasing HRT to 21 days for both cells also led to decreases in nitrates. Both these cells met human health standards displayed by Table 16. Some of the largest decreases in nitrates were seen when HLR of $\geq 2,600 \text{ m}^3$ was used for SS. Therefore, applying greater HLR to SS could potentially have led to greater decreases in nitrates. However, SSV reacted conversely from SS in terms of HLR. SSV observed a 100% nitrate removal on one occasion. This occurred when the HLR applied was $1,337 \text{ m}^3$. This was the lowest HLR applied throughout the study for SSV and observed the largest decrease in nitrates.

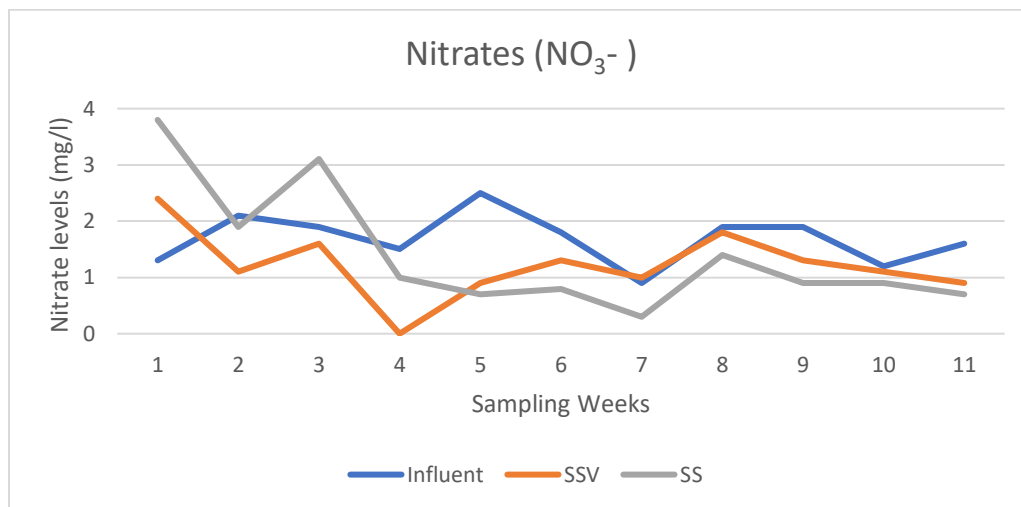


Figure 20: NO_3^- comparisons between influent, SSV and SS

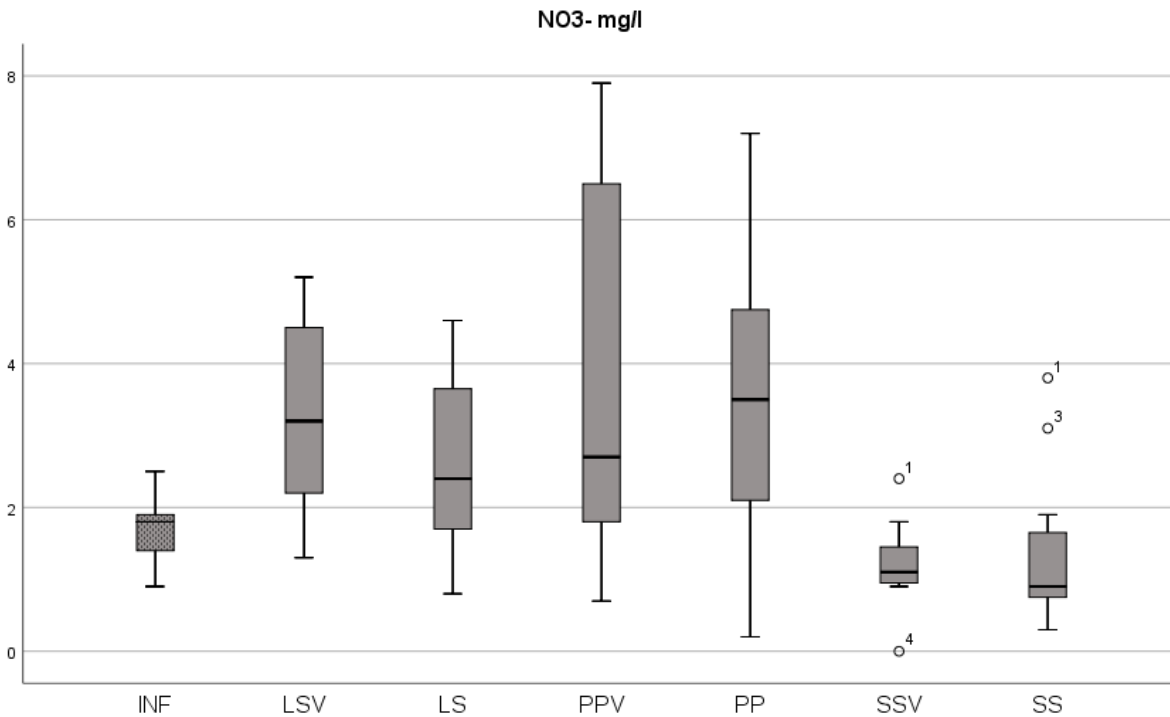


Figure 21: Box and Whisker plot for Nitrates

4.8 E. coli results for Biofiltration cells

Table 17: E. coli parameters (Mara *et al.*, 2007)

E. coli levels (CFU/100 ml)	Purpose (California Guidelines)
0	Drinking Water
≥ 235 CFU/100 ml	Foliar irrigation application
≥ 576 CFU/100 ml	Non- foliar irrigation application

As displayed by Table 17, E. coli levels should be below 235 CFU/100 ml for foliar irrigation application. However, levels up to 576 CFU/100 ml can be utilized for non- foliar irrigation purposes. Drinking water requires there to be no E. coli present (Mara *et al.*, 2007).

Table 18: E. coli values seen during sample 4.

Sample	Date	E. coli (cfu/100 ml)	%	HLR (m3)	HRT (days)
Influent	01/09/17	>2420			
LSV	01/09/17	8	-99.67%	2069	7
LS	01/09/17	6	-99.75%	3596	7
PPV	01/09/17	4	-99.83%	961	7
PP	01/09/17	>1	-99.96%	2723	7

SSV	01/09/17	4	-99.83%	1337	7
SS	01/09/17	>1	-99.96%	1003	7

Influent value was >2420 and the values for PP and SS were >1. Therefore, percentage differences should be higher.

Table 19: *E. coli* values seen during sample 7

Sample	Date	<i>E. coli</i> (cfu/100 ml)	%	HLR (m ³)	HRT (days)
Influent#2	22/09/17	241			
LSV#2	22/09/17	4	-98.34%	1974	7
LS#2	22/09/17	91	-62.24%	2243	7
PPV#2	22/09/17	299	24.07%	4079	7
PP#2	22/09/17	84	-65.15%	3875	7
SSV#2	22/09/17	>1	-99.59%	2606	7
SS#2	22/09/17	>1	-99.59%	3136	7

E. coli values for SSV and SS were <1.

Table 20: *E. coli* values seen during sample 11

Sample	Date	<i>E. coli</i> (cfu/100 ml)	%	HLR (m ³)	HRT (days)
Influent#3	20/10/17	77010			
LSV#3	20/10/17	2	-100.00%	4283	21
LS#3	20/10/17	1	-100.00%	4431	21
PPV#3	20/10/17	>2420		1876	21
PP#3	20/10/17	124	-99.84%	1808	21
SSV#3	20/10/17	1	-100.00%	1864	21
SS#3	20/10/17	<1	-100.00%	2255	21

SS value was <1

4.8.1 LSV and LS

LSV and LS displayed substantial decreases in *E. coli* levels (Tables 18,19,20) LS decreased *E. coli* levels to ≥ 8 cfu/100 ml on the three occasions where it was tested. Thus, LSV would meet the irrigation standards set by Table 17. LSV observed decreases of $\geq 98.34\%$ on HLR ranging from 1,974 m³- 4,283 m³. LS also observed large decreases in *E. coli*. However, LS did have 91 CFU/100 ml of *E. coli*. This was only a 62.24% decrease in *E. coli* levels. However, LS would still meet foliar irrigation standards set by Table 17. Higher removal efficiencies were seen with higher HLR for LS. Macrophytes have been shown to provide additional treatment

towards *E. coli* (Decamp & Warren, 2000). Thus, the plants in LSV could have potentially played role in reducing *E. coli* levels. Both cells observed the largest reduction in *E. coli* when an HRT of 21 days was applied. Decamp (2000) has stated that there is a direct relationship between *E. coli* removal and HRT.

4.8.2 PPV and PP

PPV and PP did not perform as consistently as the LSV cell. PP did manage to reduce levels below >1 cfu/100 ml and therefore was nearing drinking water levels (Tables 18, 19, 20). However, on the other two occasions tested, *E. coli* levels were still high. However, they would meet the foliar irrigation standards set in Table 17. PPV did not perform as well as PP. PPV was the only cell that increased *E. coli* values. PPV observed a 99.83% decrease in *E. coli* when an HLR of 961 m³ was applied. 961 m³ was the lowest HLR applied for any cell throughout the study. When HLR was increased PPV did not decrease *E. coli*. On the last occasion tested the PPV value was given to be $> 2,420$ cfu/100 ml, which does not meet the irrigation standards set in Table 17.

4.8.3 SSV and SS

SSV and SS displayed substantial decreases in *E. coli* levels (Tables 18,19, 20) SSV decreased *E. coli* levels to ≥ 4 cfu/100 ml on the three occasions where it was tested. Thus, SSV would meet the irrigation standards set by Table 16. SS decreased *E. coli* levels to > 1 cfu/100 ml three occasions where it was tested. Therefore, SS performed better than any other cell. Both cells met the irrigation standards set by Table 17.

4.9 Test for normality

The only distributions that are normally distributed for the Orthophosphate concentrations (mg/L) are the sites SSV and SS. And in the Ammonia concentrations, the sites Influent, LSV and LS are the only three sites that are not normally distributed.

The Influent data for both the Ammonia compound and the Orthophosphate concentrations (mg/L) are non-parametric in distribution. Thus, because the following tests are comparisons of the mean of the Influent concentrations compared with each effluent Site concentrations, the test needs to

be non-parametric to maintain not upholding the normality assumption. The Wilcoxon sign ranks test was chosen as the preferred method for this testing, as it is used for paired data, which is the case with the Water Hub influent and each effluent site concentrations (mg/L).

Table 21: Shapiro-Wilk normality tests for the NH_3 concentrations at each site. H_0 = data are normally distributed; H_A = data are not normally distributed.

Site	Test score (W)	<i>p</i>	
Influent	0.826	0.000106	***
LSV	0.764	0.00001569	***
LS	0.883	0.00399	**
PPV	0.885	0.0316	*
PP	0.787	0.000315	***
SSV	0.954	0.405	
SS	0.964	0.635	

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

Table 22: Shapiro-Wilk normality tests in Orthophosphate concentrations at each site. H_0 = data are normally distributed; H_A = data are not normally distributed

Site	Test score (W)	<i>p</i>	
Influent	0.925	0.0287	*
LSV	0.827	0.00136	**
LS	0.841	0.000953	***
PPV	0.980	0.922	
PP	0.940	0.194	
SSV	0.958	0.481	
SS	0.901	0.0715	

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

4.10 Data summary

The hypothesis method uses a paired test and was formulated in the following way:

Alternative= “two.sided”: H_0 = there is no difference in means; H_A = there is a difference in means

“greater”: H_0 = mean of the effluent > mean of influent; H_A = mean of effluent is not > than the mean of influent

“less” H_0 = mean of the effluent < mean of influent; H_A = mean of effluent is not < than the mean of influent

Table 23: Wilcoxon sign ranks test for PO_4^{3-} is used to determine whether pre-treatment is greater/less than or equal to the concentration after treatment.

Site	Test score (V)	<i>p</i>		Alternative
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Influent - LSV	388	3.866e-05	***	Greater
Influent - LS	327	0.001834	**	Greater
Influent - PPV	56	0.8355		Greater
Influent - PP	24	0.005493	**	Less
Influent - SSV	56	0.2121		Two-sided
Influent - SS	105	6.104e-05	***	Greater

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

The concentrations of PO_{43-} decreased significantly at sites LSV, LS, PPV and SS. Concentrations remained significantly unchanged at site SSV, but increased at site PPV. The sites LSV and SS display to most significant reduction in PO_{43-} concentrations as compared with other sites.

Alternative= “two.sided”: H_0 = there is no difference in means; H_A = there is a difference in means

“greater”: H_0 = mean of the effluent $>$ mean of influent; H_A = mean of effluent is not $>$ than the mean of influent

“less” H_0 = mean of the effluent $<$ mean of influent; H_A = mean of effluent is not $<$ than the mean of influent

Table 24: Wilcoxon sign ranks test for NH_3 is used to determine whether pre-treatment is greater/less than or equal to the concentration after treatment.

Site	Test score (V)	p		Alternative
Influent - LSV	551	4.773e-09	***	Greater
Influent - LS	404	1.118e-08	***	Greater
Influent - PPV	111	0.05444		Two-sided
Influent - PP	110	0.8582		Two-sided
Influent - SSV	225	6.199e-06	***	Greater
Influent - SS	199.5	4.196e-05	***	Greater

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

The concentrations of NH_3 decreased significantly at sites LSV, LS, SSV and SS, whereas these concentrations remained significantly unchanged at sites PPV and PP. The peach pips do not appear to reduce the concentration of NH_3 . The LS sites perform more significantly in reducing these concentrations as compared to the SS sites.

4.11 Discussion

The means of PO_{43-} concentrations decreased significantly in sites LSV, LS, PPV, SSV, SS; and increased in concentration at site PP. However, site LSV decreased the most significantly in PO_{43-} concentration as suggested by the Wilcoxon test. The means of the NH_3 compound decreased significantly again at all sites, but most significantly at site LSV.

The LSV site is consistent in reducing the concentrations of these PO_{43-} and NH_3 nutrients displaying large differences against the Influent yet remaining largely significant in tests against the Influent site as compared with other sites. Therefore, it is concluded that Site LSV is performing the best in cleaning/polishing the influent water and performs best when retention times are shorter, that is between 5 and 7 days. LSV performed better than all other cells, for most constituents. Thus, phytoremediation played a role in reducing contamination. Irrigation or domestic targets were met for numerous water quality parameters. However, improvement is required in certain parameters such as orthophosphate. Orthophosphate levels were not met and were too high to be diverted back into water bodies and safe for irrigating edible crops. DO levels were lowered in all the cells and ammonia levels were not reduced enough to sustain aquatic life. The quantity of water that could be treated via six biofiltration cells ranges from 12- 24 million litres of water per week. Thus, biofiltration cells have the scope to perform water conservation measures on a large scale and could potentially be utilized for irrigation.

PP and PPV did not perform consistency throughout the study. Results of this study indicate that peach pips should not be considered a suitable alternative for large and small stones. Peach pips should not be used as a substrate in biofiltration cells and should not be used as substrate for other sustainable urban drainage systems. These cells were more efficient when a retaining regime was utilized over flushing. HLR and HRT did impact some constituents and showed inverse relationships in other cells. Increasing HRT did not always result in greater removal concentrations and decreasing HLR did not always result in better water quality. However, both HLR and HRT played a role in numerous water quality parameters. Large reductions were seen under both high and low HLR and HRT for all cells and the same ranges should be applied again as seen in this study. More research is required to understand the optimum HLR and HRT for each cell. More samples need to be taken at different HLR and HRT. Achieving peak efficiency for each cell will require further testing to determine optimum conditions. However, these cells have indicated treatment efficacy at various HLR with an HRT of 7 days. According to El-Bestawy (2005) greater removal efficiencies were seen with higher HLR and shorter HRT. Thus, greater treatment

efficiencies could be reached with higher HLR. Samples should be taken every 7 days with the different HLR to determine the difference in water quality. After optimum HLR conditions were ascertained, different HRT could be applied to determine optimum HLR conditions. This study showed HRT of 7 days with flushing to be more effective than longer HRT. Shorter HRT with flushing could be applied to determine if greater treatment can be achieved.

This analysis, although quite limited did show results for which of the six cells performed the best (compared to the influent) in polishing the influent over the stipulated time period. To add more value to this analysis, it would be important to increase the sample size, collect data more accurately and more consistently as well as test for the effect of other factors such as retention times on the biofiltration process.

4.13 Limitations

There are a number of limitations to the data used in this analysis. These include:

- Small sample sizes.

The small sample size was further reduced during the data cleaning process which standardised the data. The small size of the sample increases the margin of error in results from analysing the data.

- Inconsistent data collection method

The data collection method was not consistent, and this resulted in some days missing matching observations for one or more of the biofiltration cells. As a result, to standardise the data for analysis, these missing observations were removed from the sample.

Chapter 5: Conclusion

The purpose of this study was to determine whether the biofiltration cells favourably affected water quality and which substrate would be most suited to treat contaminated surface runoff for the purpose of re-use and also for discharge into a freshwater system. The peach pip cells did not perform well during this study. However, the influent was not severely contaminated and thus even the effluent from the peach pip cells could be reused in some way. Each cell could treat between 2,000 m³- 4,000 m³ a week. This adds 12,000 m³- 24,000 m³ per week for all cells. This is a substantial amount of water and the resulting effluent could be used to grow water intensive crops. With the Western Cape's current drought, treating surface runoff should be considered a viable alternative to ensure water availability for people in the region.

Out of the three substrates, the large stones performed the best. Large stones with vegetation displayed large removals of nutrients. Phytoremediation also played a part and assisted in filtering the contaminated water. The small stones also improved water quality. However, the SSV cell leached orthophosphates and thus increased orthophosphate levels. Phosphate reducing agents should be applied to the cell, or the substrate should be excavated and reapplied. Thus, phosphate reducing techniques should be implemented in these cells to meet the standards required.

Conducting this study showed that the influent from Langrug would meet water quality guidelines for several parameters. The influent met water quality guidelines on nearly every occasion for pH, E.C and nitrates. However, guidelines were not met for orthophosphate and *E. coli*. Throughout the study the only cells that did not perform consistently were PPV and PP. SSV also increased orthophosphate concentrations throughout the study. However, SSV reduced *E. coli* bacteria each time it was tested. LSV, LS and SS reduced pollutants throughout the study and improved water quality. Irrigation standards were met for the water quality parameters tested. Overall, the cells performed well and were meeting irrigation standards for most of the parameters tested. However, if this effluent is going to be used to grow crops, then further testing on different contaminants such as heavy metals should be tested.

The peach pip cells should be replaced with large and small stones. Utilizing both substrates together could potentially increase pollutant removal as small stones performed well with nitrate

removals. The cells performed better with an HRT of 7 days with flushing, and this regime is recommended. Additional orthophosphate removal measures should be applied. This could include adding different mineral substrates such as slag or charcoal (Wang *et al.*, 2010). Plants such as kikuyu grass can remove orthophosphate and should be applied to the cells (Milandri *et al.*, 2011). Sand should also be applied for the substrate as it can reduce orthophosphate levels (De Rozari *et al.*, 2016). Additional measures should be taken to improve DO levels and ammonia concentrations so that a potential ecologically engineered system that sustains aquatic life and improves water quality can be created.

Utilizing an HRT of 7 days throughout the study resulted in water quality improvement. Different HLR regimes were utilized throughout this study and increasing HLR did decrease treatment capabilities of the biofiltration cells. However, there were points during the study where low HLR also reduced treatment efficiency. SSV saw greater reductions in *E. coli* when higher loading rates were tested. However, more research will be required to ensure that it was not an outlier. Overall HLR and HRT should be used in conjunction with each other. HRT over 7 days did not improve water quality compared to HRT over 7 days. Thus, 7 days appears to be optimal when implementing similar loading rates to those used in this study.

5.1 Recommendations

Treating polluted runoff and reusing it for human activities is one method to conserve water. Potential research questions that could be asked include: What are the optimal HLR and HRT for each biofiltration cell? This study has displayed the different HLRs and HRTs that were utilized during this study. Specifically:

- Monitoring various HLRs under the same HRT. A greater variation of HLR's should be monitored for this study. Through greater data collection and analyses an optimum HLR could be determined for each cell.
- Utilizing different HRTs. This study only used HRT's of 7 days and 14 days. Analysis should be conducted daily. This will assist in determining optimum HRT for each cell.
- Analysing different water quality parameters. Other parameters such as BOD, COD and

heavy metal tests should be tested.

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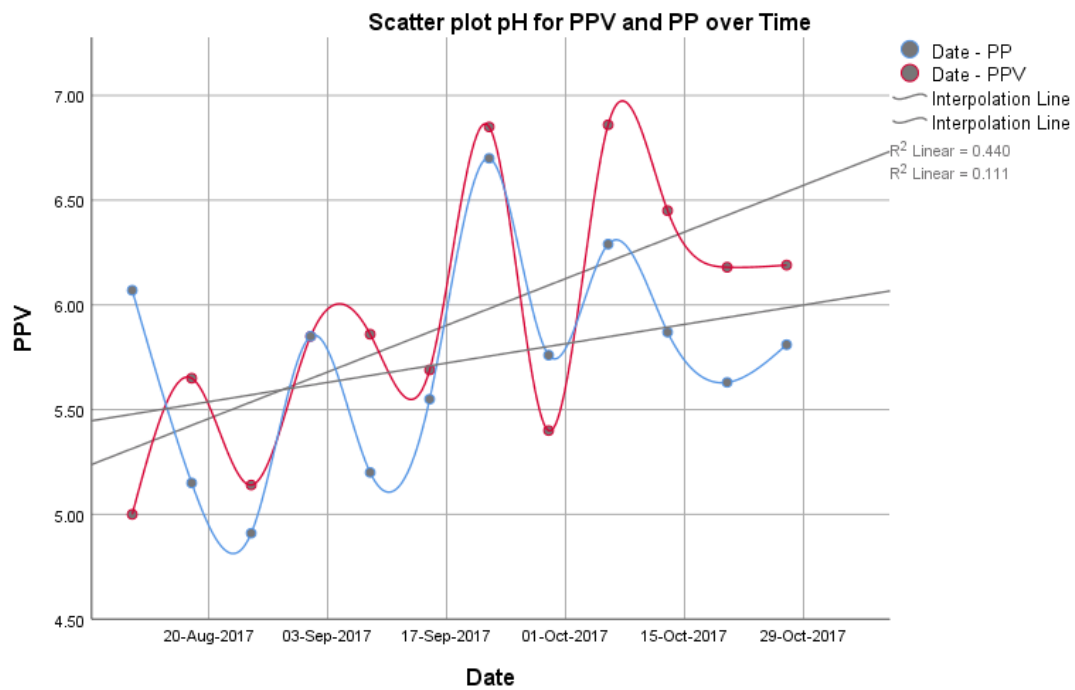
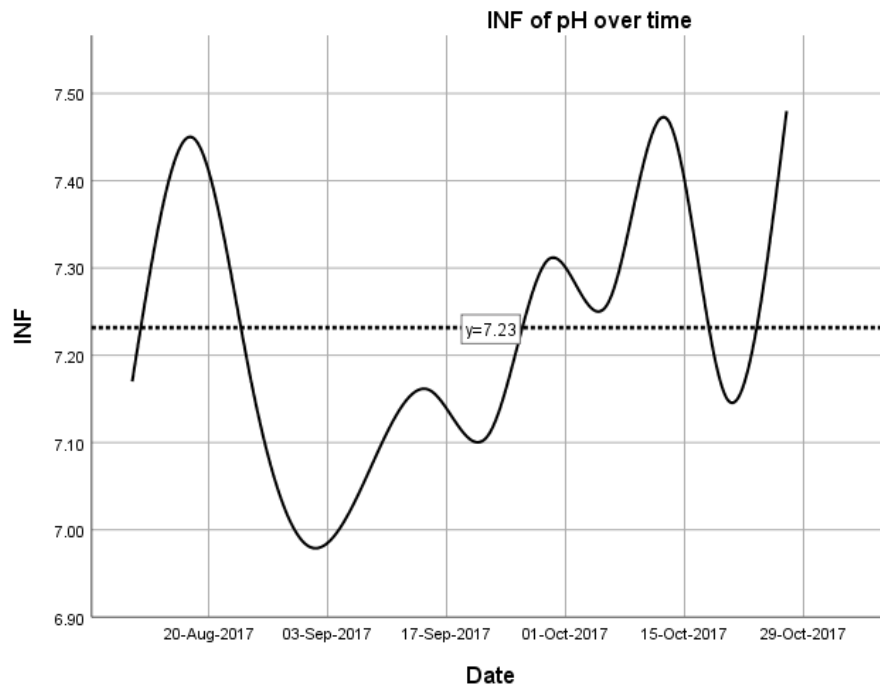
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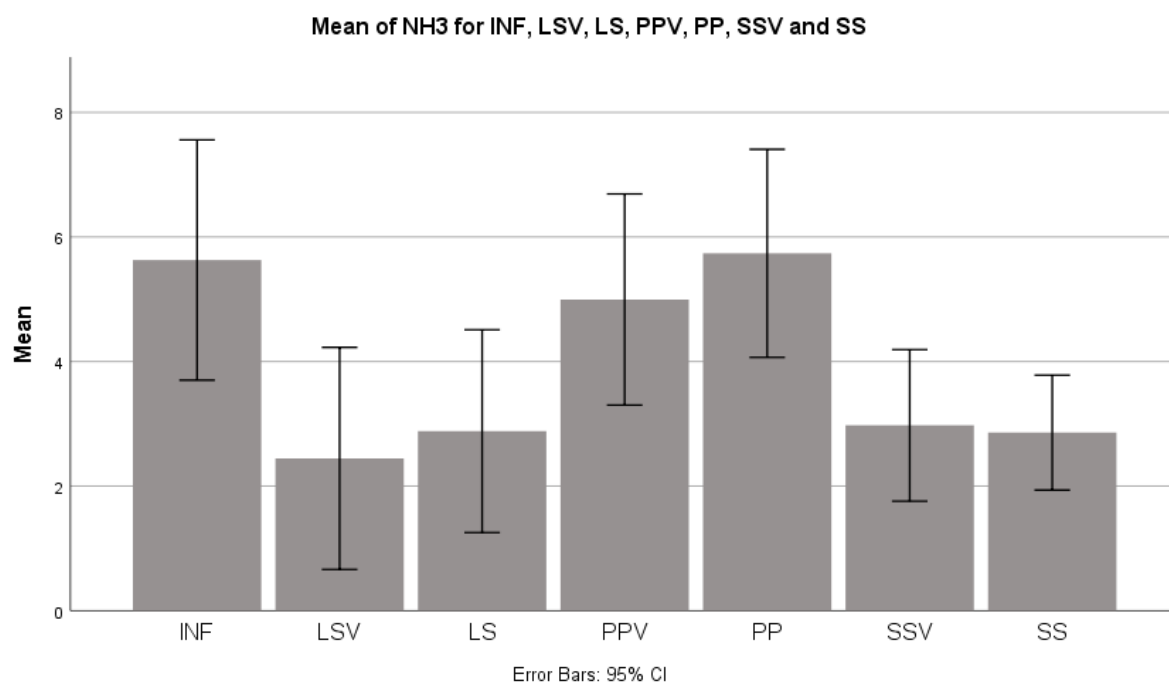
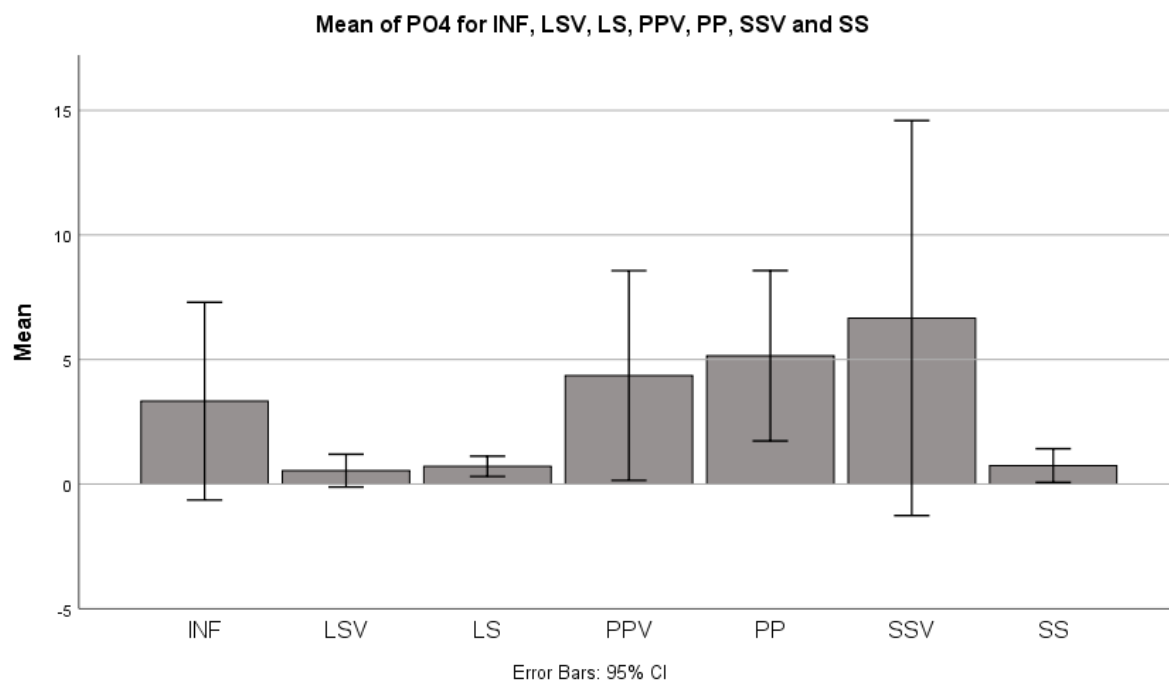
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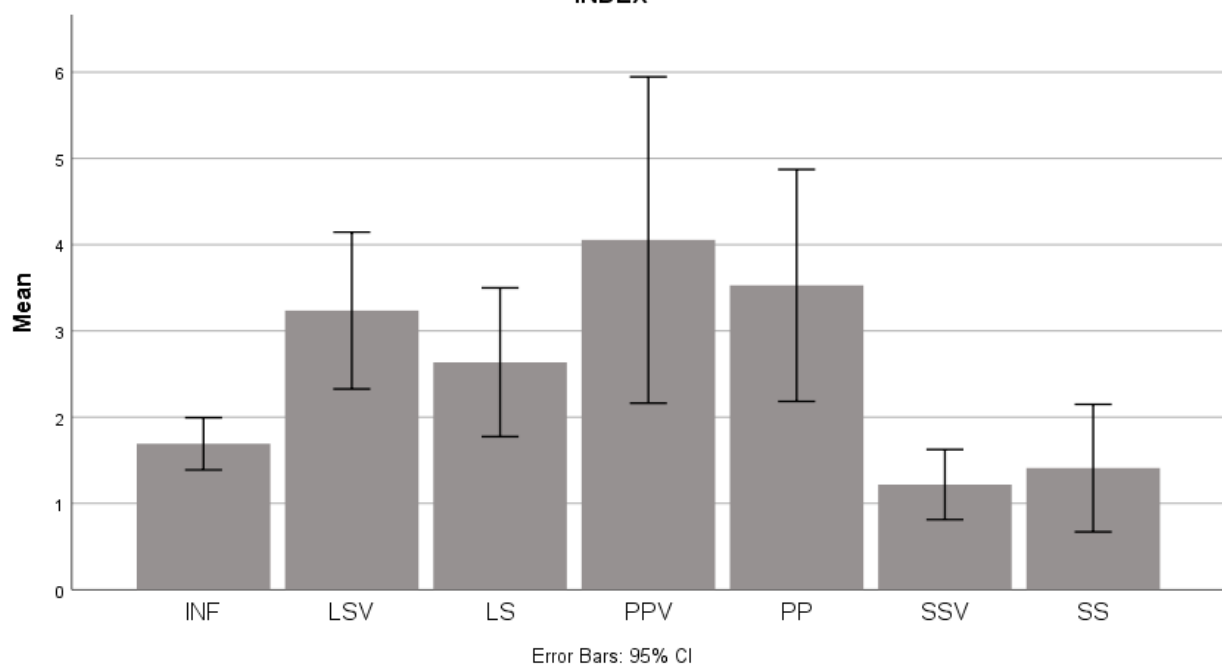
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Appendix

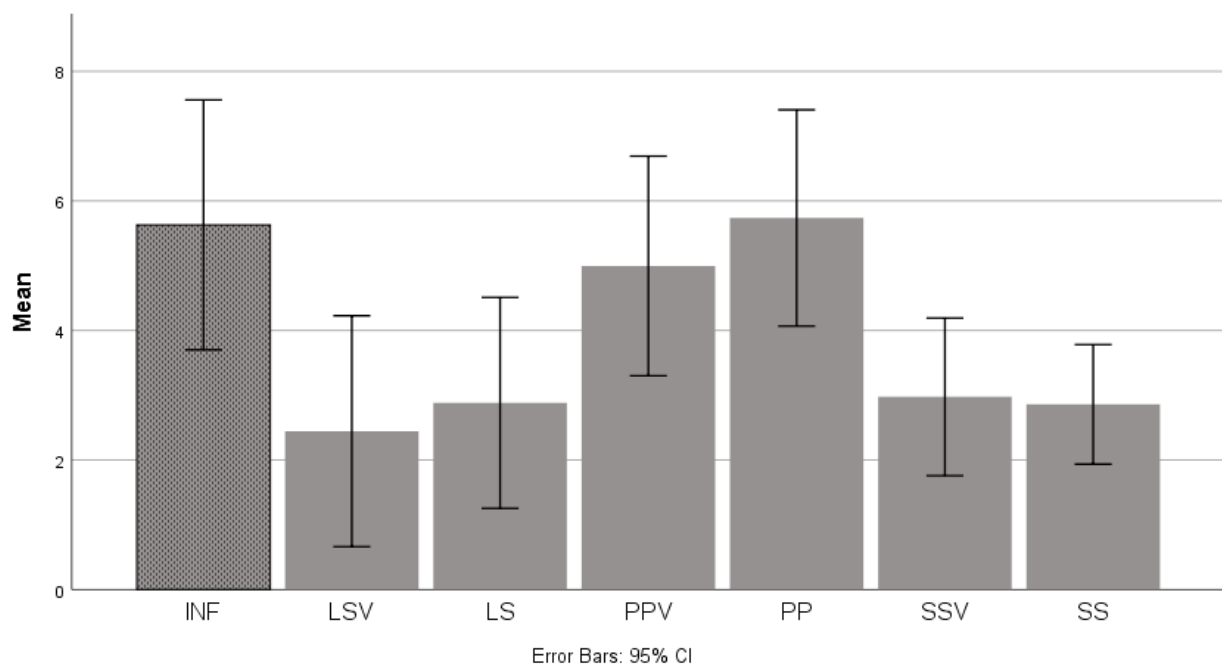




Simple Bar Mean of INF, Mean of LSV, Mean of LS, Mean of PPV, Mean of PP, Mean of SSV, Mean of SS by INDEX



Mean of D.O. for INF, LSV, LS, PPV, PP, SSV and SS



Tests of Normality for D.O.

	Kolmogorov-Smirnov ^a			Shapiro-Wilk		
	Statisti c	df	Sig.	Statisti c	df	Sig.
INF	.290	11	.010	.749	11	.002
LS V	.252	11	.049	.845	11	.037
LS	.156	11	.200*	.918	11	.304 _a
PP V	.139	11	.200*	.965	11	.835 _a
PP	.171	11	.200*	.928	11	.393 _a
SS V	.131	11	.200*	.929	11	.401 _a
SS	.199	11	.200*	.942	11	.546 _a

*. This is a lower bound of the true significance.

a. Test for normality >0.05 therefore reject the null hypothesis for normality in distribution for these variables

Tests of Normality for NH3

	Kolmogorov-Smirnov ^a			Shapiro-Wilk		
	Statisti c	df	Sig.	Statisti c	df	Sig.
INF	.290	11	.010	.749	11	.002
LS V	.252	11	.049	.845	11	.037
LS	.156	11	.200*	.918	11	.304
PP V	.139	11	.200*	.965	11	.835
PP	.171	11	.200*	.928	11	.393
SS V	.131	11	.200*	.929	11	.401
SS	.199	11	.200*	.942	11	.546

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

Tests of Normality for PO₄₃

	Kolmogorov-Smirnov ^a			Shapiro-Wilk		
	Statisti c	df	Sig.	Statisti c	df	Sig.

INF	.441	11	.000	.462	11	.000
LS V	.336	11	.001	.560	11	.000
LS	.345	11	.001	.745	11	.002
PP V	.432	11	.000	.516	11	.000
PP	.312	11	.004	.662	11	.000
SS V	.394	11	.000	.556	11	.000
SS	.320	11	.002	.686	11	.000

a. Lilliefors Significance Correction

Tests of Normality for NO3

	Kolmogorov-Smirnov ^a			Shapiro-Wilk		
	Statistic	df	Sig.	Statistic	df	Sig.
INF	.141	11	.200*	.977	11	.950
LS V	.188	11	.200*	.930	11	.411
LS	.118	11	.200*	.954	11	.700
PP V	.230	11	.107	.866	11	.068
PP	.123	11	.200*	.982	11	.977
SS V	.208	11	.198	.944	11	.565
SS	.281	11	.015	.804	11	.011

*. This is a lower bound of the true significance.

a. Lilliefors Significance Correction

Test Statistics^a for NO3-

	LSV - INF	LS - INF	PPV - INF	PP - INF	SSV - INF	SS - INF
Z	-3.059 _b	- 3.059 _b	-1.201 _b	-.445 _c	-2.845 _b	- 2.934 _b
Asymp. Sig. (2-tailed)	.002	.002	.230	.656	.004	.003

a. Wilcoxon Signed Ranks Test

b. Based on positive ranks.

c. Based on negative ranks.

Test Statistics^a for PO₄₃

	LSV - INF	LS - INF	PPV - INF	PP - INF	SSV - INF	SS - INF
Z	-3.059 _b	- 3.059 _b	-1.201 _b	-.445 _c	-2.845 _b	- 2.934 _b
Asymp. Sig. (2-tailed)	.002	.002	.230	.656	.004	.003

a. Wilcoxon Signed Ranks Test

b. Based on positive ranks.

c. Based on negative ranks.

Test Statistics^a for NH₃

	LSV - INF	LS - INF	PPV - INF	PP - INF	SSV - INF	SS - INF
Z	-3.059 _b	- 3.059 _b	-1.201 _b	-.445 _c	-2.845 _b	- 2.934 _b
Asymp. Sig. (2-tailed)	.002	.002	.230	.656	.004	.003

a. Wilcoxon Signed Ranks Test

b. Based on positive ranks.

c. Based on negative ranks.

Test Statistics^a for EC

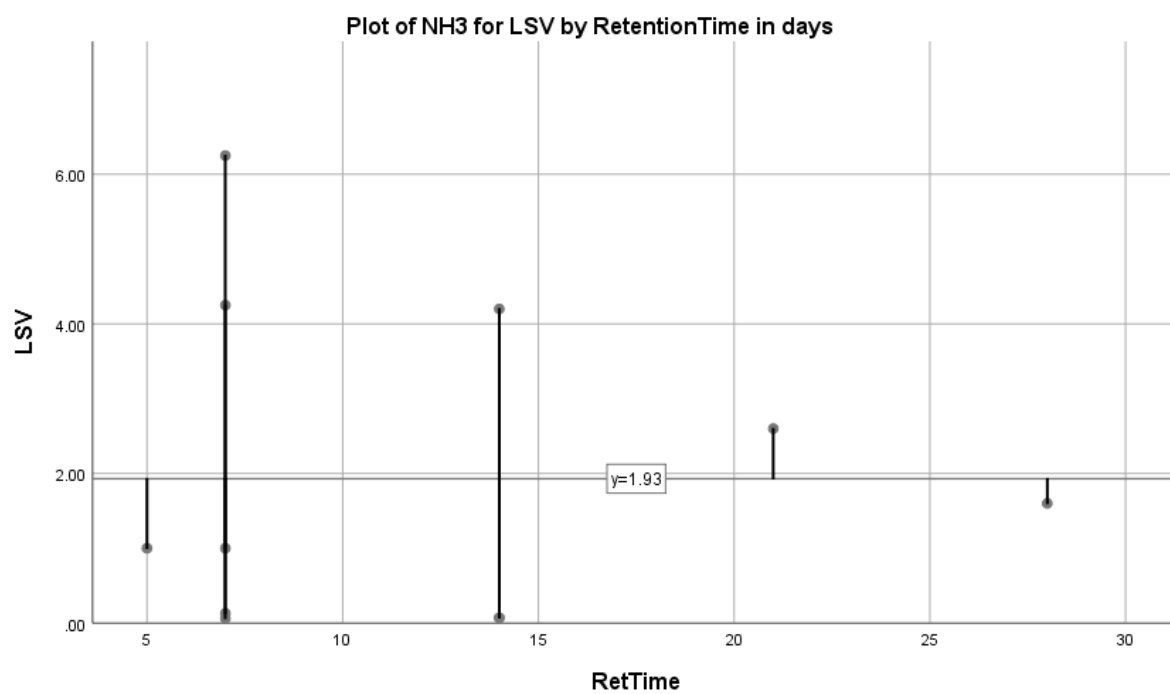
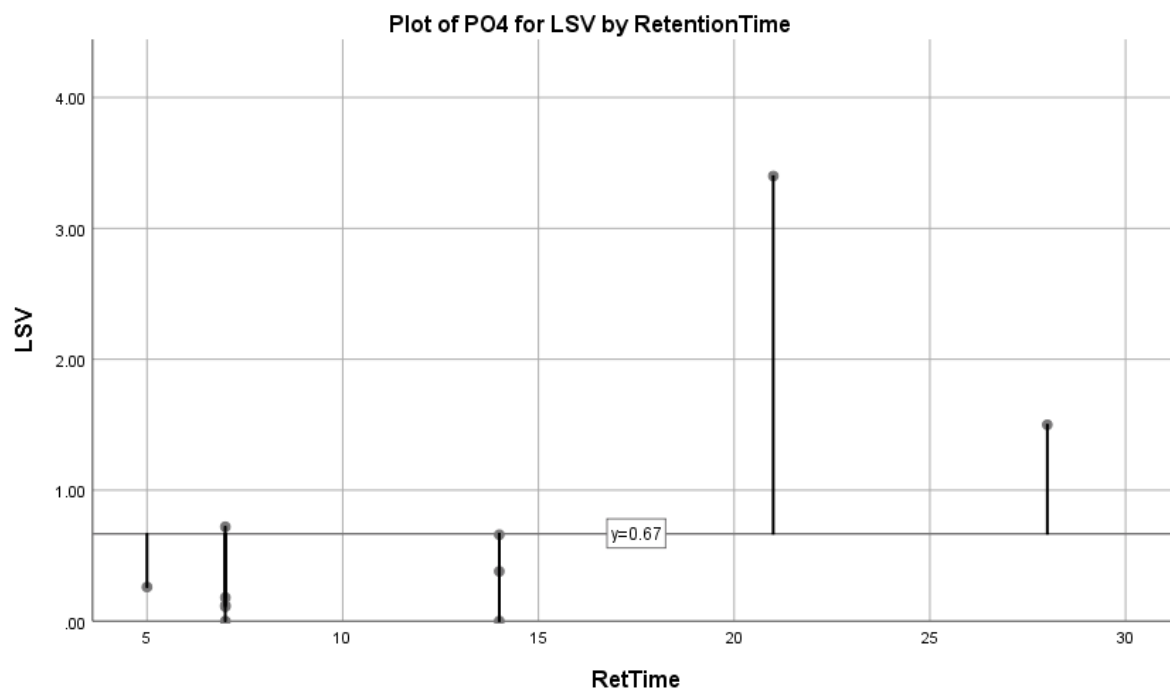
	LSV - INF	LS - INF	PPV - INF	PP - INF	SSV - INF	SS - INF
Z	-3.059 _b	- 3.059 _b	-1.201 _b	-.445 _c	-2.845 _b	- 2.934 _b
Asymp. Sig. (2-tailed)	.002	.002	.230	.656	.004	.003

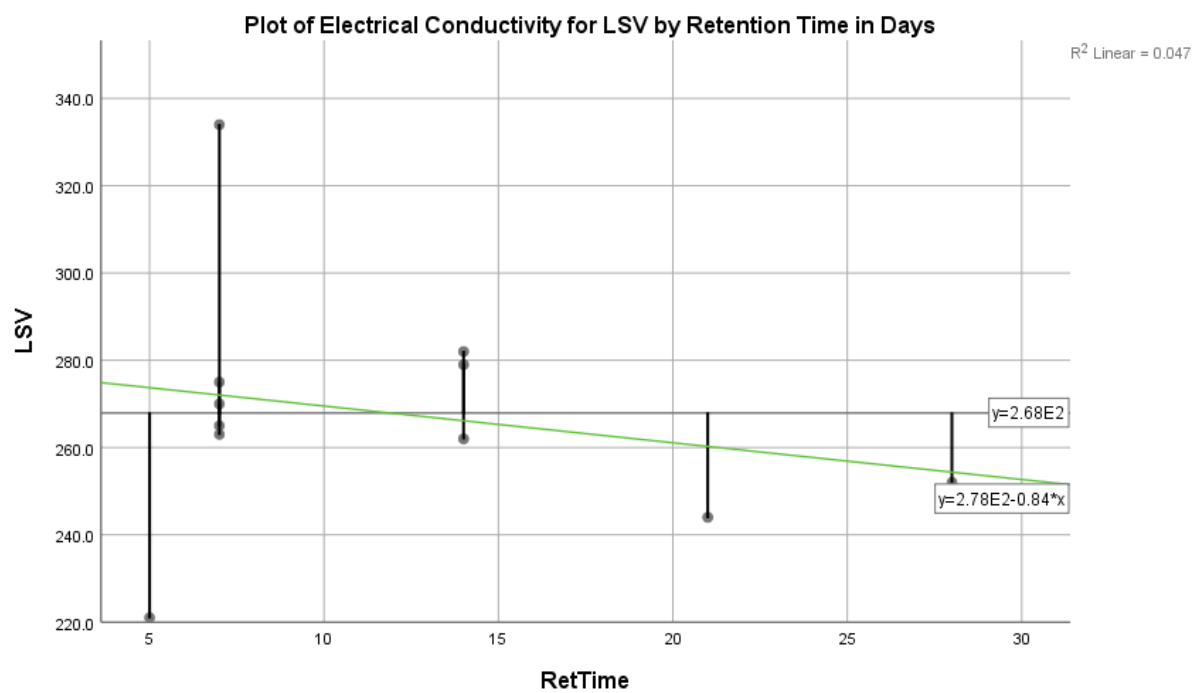
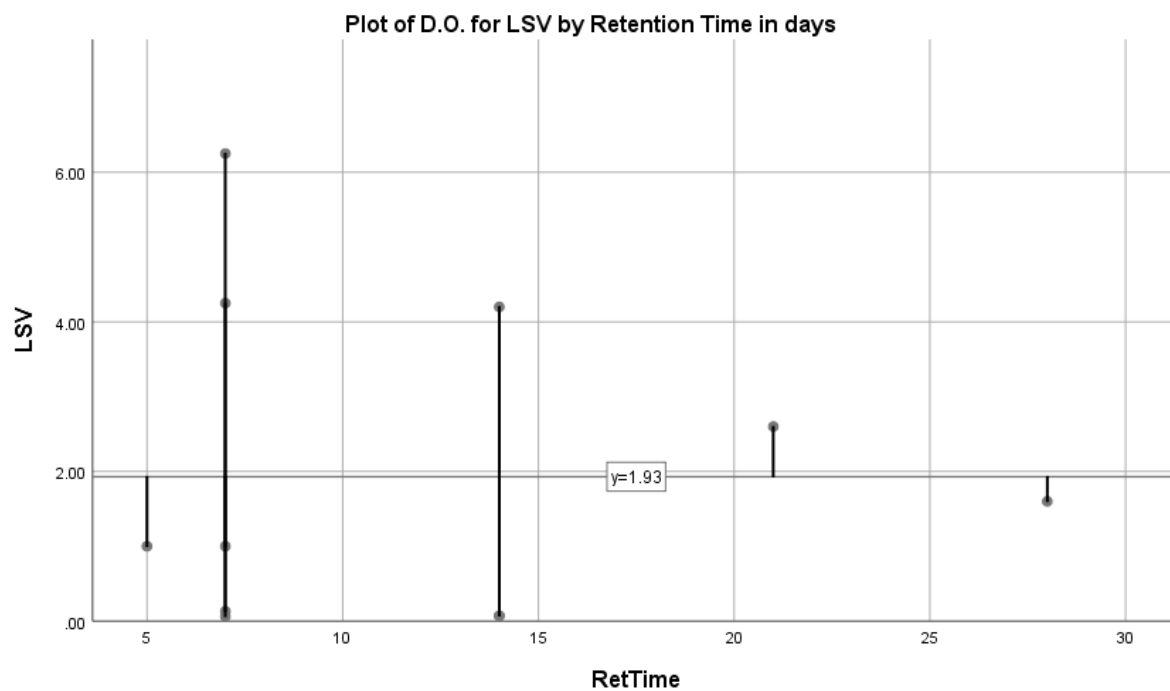
a. Wilcoxon Signed Ranks Test

b. Based on positive ranks.

c. Based on negative ranks.

Retention time





LSV

Ammonia:

F critical value: 3.78704354 > 1.288395974 (F-value) at an HRT of 7 days

F critical value: 9.276628153 > 7.400023628 (F-value) at an HLR < 2,000 m³

F critical value: 0.052631579 > 0.047416273 (F-value) at an HLR > 2,600 m³

P-value: 0.012756239, Standard Error: 2.089491303 at an HRT of 7 days

P- value: 0.041458827, Standard Error: 0.419912232 at an HLR 2000-2,600 m³

Nitrate:

F critical value: 9.276628153 > 7.11875 (F-value) at an HRT over 7 days

F critical value: 9.276628153 > 5.264663805 (F-value) at an HLR < 2,000 m³

Nitrite:

F critical value: 9.276628153 > 2.825863724 (F-value) at HLR 2,000-2,600 m³

Orthophosphate:

F critical value: 9.276628153 > 6.276247278 (F-value) at HLR < 2,000

p- value: 0.020099428, Standard Error: 1.660930975 at HLR < 2,000 m³

DO:

F critical value: 9.276628153 > 4.435483871 (F-value) at HRT over 7 days

F critical value: 9.276628153 > 2.195121951 (F-value) at HLR 2,000-2,600 m³

F critical value: 19 > 1.895522388 (F-value) at HLR > 2,600 m³

pH:

F critical value: 0.264058226 > 0.226078261 (F-value) at HRT of 7 days

F critical value: 9.276628153 > 2.180851064 (F-value) at HRT over 7 days

F critical value: 9.276628153 > 4.090999011 (F-value) at HLR < 2,000 m³

F critical value: 19 > 1.550458716 (F-value) at HLR > 2,600 m³

ORP:

F critical value: 9.276628153 > 2.365725542 (F- value) at an HRT over 7 days

F critical value: 0.107797789 > 0.037859206 (F- value) at HLR 2,000-2,600 m³

p-value: 0.008401662, Standard Error: 34.74654945 at an HRT of 7 days

E.C:

F critical value: 0.107797789 > 0.039108486 (F-value) at HLR < 2,000 m³

F critical value: 9.276628153 > 1.001356636 (F-value) at HLR 2,000-2,600 m³

F critical value: 19 > 3.041307308 (F-value) at HLR > 2,600 m³

TDS:

F critical: 9.276628153 > 1.421153396 (F-value) at HLR < 2,000 m³

F critical: 19 > 3.49508023 (F-value) at HLR > 2,600 m³

LS

Ammonia:

F critical: 3.78704354 > 1.621312859 (F-value) at HRT of 7 days

F critical: 9.276628153 > 1.738892051 (F-value) at HLR < 2,200 m³

p-values: 0.003927834, Standard Error: 1.731472309 at HRT of 7 days

Nitrate:

F critical: 3.78704354 > 2.8952923 (F-value) at HRT of 7 days

F critical: 0.107797789 > 0.09057872 (F-value) at HRT over 7 days

F critical: 9.276628153 > 1.417833713 (F-value) at HLR < 2,200 m³

F critical: 9.276628153 > 4.914994502 (F-value) at HLR 2,200-3,600 m³

p-value: 0.012048673, Standard Error: 0.305742619 at HRT of 7 days

p-value: 0.022619891, Standard Error: 0.197689647 at HLR < 2,200 m³

Nitrite:

F critical: 3.78704354 > 2.8952923 (F-value) at HRT of 7 days

F critical: 0.107797789 > 0.09057872 (F-value) at HRT over 7 days

F critical: 9.276628153 > 1.417833713 (F-value) at HLR < 2,200 m³

F critical: 9.276628153 > 4.914994502 (F-value) at HLR 2,200-3,600 m³

Orthophosphate:

F critical: 9.276628153 > 2.601398601 at HLR 2,200-3,600 m³

p-value: 0.008556173, Standard Error: 1.109095098 at HLR < 2,200 m³

DO:

F critical: 9.276628153 > 8.225 (F-value) at HLR 2,200-3,600 m³

F critical: 19 > 1.194842407 (F-value) at HLR > 3,600 m³

pH:

F critical: 9.276628153 > 5.331040413 (F-value) at HLR 2,200-3,600 m³

ORP:

F critical: 0.264058226 > 0.13367798 (F-value) at HRT of 7 days

F critical: 0.107797789 > 0.034748645 (F-value) at HLR < 2,200 m³

F critical: 19 > 14.22274882 (F-value) at HLR > 3,600 m³

p-value: 0.003006159, Standard Error: 29.46349812 at HRT of 7 days

E.C

F critical: 3.78704354 > 3.06973176 (F-value) at HRT of 7 days

F critical: 9.276628153 > 1.510117493 (F-value) at HRT over 7 days

F critical: 9.276628153 > 1.067684628 (F-value) at HLR < 2,200 m³

F critical: 9.276628153 > 5.720068611 (F-value) at HLR 2,200-3,600 m³
F critical: 19 > 3.778677463 (F-value) at HLR > 3,600 m³

TDS

F critical: 0.107797789 > 0.01957786 (F-value) at HRT over 7 days
F critical: 9.276628153 > 1.777526456 (F-value) at HLR < 2,200 m³

PPV

Ammonia:

F critical: 3.78704354 > 1.325868801 (F-value) at HRT of 7 days
F critical: 19 > 2.338129496 (F-value) at HLR > 4,100 m³

p-value: 0.037216668, Standard Error: 2.467585639 at HRT of 7 days

Nitrate:

F critical: 19 > 1.025316456 (F-value) at HLR > 4,100 m³

Nitrite:

F critical: 3.78704354 > 2.272720757 (F-value) at HRT of 7 days
F critical: 9.276628153 > 5.13129928 (F-value) at HLR < 4,100 m³

Orthophosphate:

F critical: 0.264058226 > 0.010080911 (F-value) at HRT of 7 days
F critical: 9.276628153 > 1.005471601 (F-value) at HLR < 2,000 m³
F critical: 9.276628153 > 1.556863805 (F-value) at HLR < 4,100 m³

p-value: 0.019064592, Standard Error: 0.30269963 at HLR > 4,100 m³

DO:

F critical: 9.276628153 > 1.968911917 at HLR < 2,000 m³
F critical: 9.276628153 > 5.023255814 at HLR < 4,100 m³

pH:

F critical: 0.264058226 > 0.043790332 (F-value) at HRT of 7 days
F critical: 0.107797789 > 0.045816194 (F-value) at HLR < 4,100 m³
F critical: 0.052631579 > 0.024901704 (F-value) at HLR > 4,100 m³

ORP:

F critical: 0.264058226 > 0.093649998 (F-value) at HRT of 7 days
F critical: 9.276628153 > 4.198551287 (F-value) at HLR < 4,100 m³

E.C:

F critical: 3.78704354 > 1.008172509 (F-value) at HRT of 7 days

p-value: 0.015752448, Standard Error: 35.51807929 at HRT of 7 days
p-value: 0.017048837, Standard Error: 10.59749472 at HRT over 7 days
p-value: 0.025823232, Standard Error: 16.65224607 at HLR < 2,000 m³
p-value: 0.006821689, Standard Error: 6.350852961 at HLR < 4,100 m³

TDS:

F critical: 0.107797789 > 0.057909762 (F-value) at HRT over 7 days

p-value: 0.009845059, Standard Error: 3.251114258 at HRT over 7 days
p-value: 0.034061186, Standard Error: 9.064706575 at HLR < 2,000 m³

PP

Ammonia:

F critical: 3.78704354 > 1.368837035 (F-value) at HRT of 7 days
F critical: 9.276628153 > 1.463065914 (F-value) at HLR < 3,000 m³
F critical: 19 > 1.215827338 (F-value) at HLR > 4,000 m³

p-value: 0.004695977, Standard Error: 1.523069467 at HRT of 7 days
p-value: 0.02824288, Standard Error: 0.362812273 at HRT over 7 days

Nitrate:

F critical: 9.276628153 > 8.95 (F-value) at HRT over 7 days
F critical: 19 > 3.197530864 (F-value) at HLR < 4,000 m³

p-value: 0.02262507, Standard Error: 1.545516318 at HRT of 7 days
p-value: 0.024161773, Standard Error: 0.613731755 at HLR < 4,000 m³

Nitrite:

F critical: 3.78704354 > 2.633947793 (F-value) at HRT of 7 days
F critical: 9.276628153 > 2.756225426 (F-value) at HLR < 3,000 m³
F critical: 9.276628153 > 3.62346106 (F-value) at HLR < 4,000 m³
F critical: 19 > 3.650231371 (F-value) at HLR > 4,000 m³

p-value: 0.025785495, Standard Error: 0.035082339 at HRT of 7 days
p-value: 0.007752817, Standard Error: 0.007471804 at HLR < 4,000 m³

Orthophosphate:

F critical: 9.276628153 > 2.648970551 (F-value) at HLR < 4,000 m³

DO:

F critical: 9.276628153 > 1.397712834 (F-value) at HRT over 7 days
F critical: 9.276628153 > 2.373626374 (F-value) at HLR < 4,000 m³
F critical: 19 > 4.191860465 (F-value) at HLR > 4,000 m³

pH:

F critical: 0.264058226 > 0.057066195 at HRT of 7 days
F critical: 9.276628153 > 1.518518519 at HRT over 7 days
F critical: 0.107797789 > 0.048412567 at HLR < 4,000 m³

p-value: 0.04663464, Standard Error: 0.055447624, HRT over 7 days

ORP:

p-value: 0.040314534, Standard Error: 5.965009561 at HRT over 7 days

E.C:

F critical: 3.78704354 > 2.017700654 (F-value) at HRT of 7 days
F critical: 9.276628153 > 2.271727826 (F-value) at HLR < 3,000 m³
F critical: 19 > 9.99852071 (F-value) at HLR > 4,000 m³

p-value: 0.054620563, Standard Error: 9.455294271 at HLR < 3,000 m³

TDS:

F critical: 0.107797789 > 0.023900603 (F-value) at HRT over 7 days
F critical: 19 > 3.221885962 (F-value) at HLR > 4,000 m³

4.12.5 SSV**Ammonia**

F critical: 3.78704354 > 2.572862607 (F-value) at HRT of 7 days
F critical: 19 > 1.123134328 (F-value) at HRT over 7 days
F critical: 5.050329058 > 1.929313929 (F-value) at HLR > 2,300 m³

Nitrate

F critical: 19 > 2.333333333 (F-value) at HRT over 7 days
F critical: 5.050329058 > 3.155844156 (F-value) at HLR > 2,300 m³

Nitrite

F critical: 19 > 4.48853211 (F-value) at HRT over 7 days

p-value: 0.049854849, Standard Error: 0.01624173 at HLR > 2,300 m³

Orthophosphate

F critical: 9.276628153 > 3.868930424 (F-value) at HLR < 2,300
F critical: 5.050329058 > 1.455181277 (F-value) at HLR > 2,300

DO

F critical: 3.78704354 > 2.392152877 (F-value) at HRT of 7 days
F critical: 19 > 13.85714286 (F-value) at HRT over 7 days
F critical: 5.050329058 > 4.402719665 (F-value) at HLR > 2,300

pH

F critical: 3.78704354>1.167112811 (F-value) at HRT of 7 days

F critical: 19>1.25220681 (F-value) at HRT over 7 days

F critical: 9.276628153>1.352066607 (F-value) at HLR< 2,300 m³

F critical: 5.050329058>1.62546262 (F-value) at HLR> 2,300 m³

ORP

F critical: 0.052631579>0.023717508 (F-value) at HRT over 7 days

F critical: 5.050329058>1.756993631 (F-value) at HLR> 2,300 m³

p-value: 0.002431641, Standard Error: 28.46943434 at HRT of 7 days

E.C

F critical: 9.276628153>1.958919217 (F-value) at HLR< 2,300 m³

TDS

F critical: 9.276628153>6.092019988 (F-value) at HLR< 2,300 m³

p-value: 0.016130918, Standard Error: 206.9509612 at HRT of 7 days

SS**Ammonia**

F critical: 19>2.090277778 (F-value) at HRT over 7 days

Nitrate

F critical: 0.264058226>0.164792234 (F-value) at HRT of 7 days

F critical: 19>9.333333333 (F-value) at HRT over 7 days

F critical: 0.107797789>0.065232975 (F-value) at HLR< 2,800 m³

F critical: 5.050329058>2.557894737 (F-value) at HLR> 2,800 m³

Nitrite

F critical: 19>11.37790698 (F-value) at HRT over 7 days

F critical: 5.050329058>1.513670946 (F-value) at HLR> 2,800 m³

p-value: 0.024277471, Standard Error: 0.056408143 at HRT of 7 days

Orthophosphate

p-value: 0.003078384, Standard Error: 0.361054992 at HRT of 7 days

p-value: 0.000361655, Standard Error: 0.328319043 at HLR< 2,800 m³

DO

F critical: 3.78704354>1.783148962 (F-value) at HRT of 7 days

F critical: 19>2.917293233 (F-value) at HRT over 7 days

pH

F critical: 19 > 9.194444444 (F-value) at HRT over 7 days

p-value: 0.020598089, Standard Error: 0.048139097 at HLR < 2,800 m³

ORP

F critical: 9.276628153 > 1.464887413 (F-value) at HLR < 2,800 m³

p-value: 0.002348444, standard error: 28.30928719 at HRT of 7 days

E.C

F critical: 19 > 1.971493729 (F-value) at HRT over 7 days

F critical: 9.276628153 > 3.002688172 (F-value) at HLR < 2,800 m³

TDS

F critical: 5.050329058 > 1.514130218 (F-value) at HLR > 2,800 m³

p-value: 0.001605469, Standard Error: 142.9856081 at HRT of 7 days

